Ecosystem monitoring of subtidal reefs in different management zones of the Jurien Bay Marine Park 1999–2007

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Summary

Floral and faunal communities associated with shallow reefs in the Jurien Bay Marine Park were investigated using underwater visual census methods on six occasions from October 1999 to November 2007. Densities of fishes, mobile macro-invertebrates, sessile invertebrates and macro-algae were quantified at a total of 42 sites.

The magnitude of changes in density at zone locations between years was generally considerably less than spatial variability between zone locations, although some species and community metrics examined showed consistent increases or decreases in numbers over the survey period.

The biotic community at different sites showed a major biotic subdivision between reefs located within a kilometre of the coastline and outer lagoonal reefs. Inshore reefs were characterised by sheltered conditions, water with moderate turbidity, and by macro-algae such as *Dictymenia sonderi* and *Neurymenia fraxinifolia*. Outer reefs were characterised by clear water, wave-exposed conditions, the kelp *Ecklonia radiata*, and the red seaweeds *Pterocladia lucida* and *Hennedya crispa*.

Outer reefs were largely homogeneous with respect to associated plants and animals, whereas inshore reefs showed considerable variation between sites and between management zone locations. Scientific reference zones, in which rock lobster fishing is permitted, largely include outer reef systems. As a consequence, scientific reference zones possess a biota that differs little between zone locations. By contrast, sanctuary zones, where all forms of fishing are prohibited, include only shallow reefs, hence possess a biota that varies greatly between locations, and also differs from the biota encountered in scientific reference zones.

The lack of overlap in reef communities between sanctuary zones and scientific reference zones complicated analysis of effects of restrictions on fishing that were enacted in these two zone types in December 2005. Regardless, very few observable ecological changes associated with new fishing restrictions were identified. The strongest relationships associated with fishing restrictions were found in correlation analysis when comparing the level of change from the periods before and after fishing restrictions with areal extent of protected zone. In this analysis, the mean size of breaksea cod (*Epinephelides armatus*) and silver trevally (*Caranx georgianus*) was found to increase significantly in the larger protected zones, as did the abundance of large (>250 mm) exploited fishes as a group and also dhufish (*Glaucosoma hebraicum*).

We consider that the trends in breaksea cod size and numbers of exploited fishes are probably real, given that adequate data were available for statistical tests and trends were consistent between years. However, the trends in silver trevally size and dhufish abundance relate to highly patchy data, with the significant correlations best regarded as flags for confirmation or otherwise through the longer term.

Several management recommendations arose from the study:

Sanctuary zones should be extended to the outer reef area. Extra sites should then be added to the JBMP reef monitoring program to allow improved evaluation of effects of

fishing restrictions, particularly with respect to assessment of the ecosystem role of rock lobsters.

Other than for an extension of sanctuary zones to the offshore region, the system of protected zones in the JBMP should be maintained with as few changes to regulations and zone boundaries as possible through the long term. This is particularly important for zones that include sites surveyed as part of the long-term JBMP reef monitoring program.

Surveys of fishes and mobile invertebrates should be repeated on an annual basis, and surveys of plant assemblages be conducted on a biennial basis, for at least five years from the time of enforcement of fishing restrictions in 2005. The frequency of surveys should be reviewed in 2010 to assess whether a longer period between surveys is warranted on grounds of cost-effectiveness. Monitoring should nevertheless continue through the longer term at least until biotic changes associated with MPA protection stabilise, probably longer given the unique value of the data set in tracking ecological effects of climate change.

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1. Introduction

Marine conservation planning has developed rapidly over the past two decades, evolving from an era when marine protected areas (MPAs) barely existed, and were largely declared on a *ad hoc* basis (with the notable exception of the Great Barrier Reef Marine Park), to detailed systematic planning of networks today. In Australia, MPA developments in different state, territory and Commonwealth waters now form part of a national system of representative marine protected areas (NRSMPAs) (ANZECC 1999).

MPAs can potentially provide a wide range of benefits to human society, including enhanced conservation of biodiversity—the primary goal of the NRSMPA. For example, MPAs can be used in fisheries management to conserve critical habitats and protect spawner biomass. MPAs also act as reference areas for assessing acute or chronic impacts caused by anthropogenic activities, and for assessing the success of conservation and fisheries management strategies, both for single species and ecosystems (Roberts *et al.* 2001; Russ 2002; Ward *et al.* 2001).

To assess the effectiveness of MPAs as a management strategy, field monitoring programs are necessary to identify changes that accompany declaration of MPAs, and to assess if these changes result from protection rather than natural variation. In this way, monitoring programs can inform MPA planners on which management strategies and design principles are most effective in achieving desired outcomes.

Sound scientific design for MPA monitoring programs requires replicated surveys both within and adjacent to protected zones, ideally from the period prior to declaration of MPAs (Willis *et al.* 2003). As much as practicable, control sites should have similar habitat, oceanographic and geographic characteristics to the protected or treatment areas. Surveys should also be repeated multiple times both before the reserves are restricted to fishing, and then for a biologically meaningful period following protection. Through the use of time-series sampling designs, the effectiveness of various levels of protection can be distinguished from more general long-term trends that are coincidental to MPA management strategies.

Grants from the Australian Research Council (ARC), Fisheries Research and Development Corporation (FRDC), and assistance from the Commonwealth Government and various State governments have enabled the Tasmanian Aquaculture and Fisheries Institute (TAFI) to undertake baseline and follow-up surveys in a range of proposed Australian temperate MPAs. Study sites have been established in Western Australia, South Australia, Victoria, Tasmania and New South Wales. In each state a common methodology was used that allows for comparison of results between differing locations, MPA designs and management strategies. This information can be used to critique current plans and assist future planning.

In Western Australia, monitoring surveys have been conducted since 1999 within the Jurien Bay Marine Park (JBMP), located 250 km north of Perth in the Central West Coast marine bioregion (Interim Marine and Coastal Regionalisation for Australia Technical Group 1998). The JBMP is centred around the towns of Jurien and Cervantes, comprises an area of 824 km^2, and extends along approximately 80 km of coastline from the shore to the limits of State jurisdiction at an offshore distance of three nautical miles (ca. 5 km; Fig. 1). The MPA is characterised by an extensive offshore development of

limestone pavement, structured reef, seagrass beds, sand banks and islands that provide a protective barrier from the prevailing swells and seas. Wave height generally declines substantially eastward of a series of reefs running north-south at a distance of 5-7 km offshore. The inner three kilometres of coastal waters essentially form a protected lagoon with water depths generally <5 m. Isolated structured reefs outcrop in this area from sand and seagrass.

The JBMP was declared on 26 August 2003 and is zoned for multiple use, with six categories of management zone affording different levels of protection (Department of Conservation and Land Management 2005). Restrictions on fishing in the various zones were enacted on 23 Dec 2005. The most highly protected zones within the MPA comprise ten sanctuary zone locations (3.7% of total area; Table 1). Three scientific reference zone locations are also highly protected (17% of total area; Table 2), while 77% of the marine park is zoned for general use, permitting most forms of fishing. Restrictions associated with these zone types are as follows:

1. Sanctuary Zones provide the highest level of protection for vulnerable or specially-protected species and protect representative habitats from human disturbance. Passive activities are permitted and extractive activities are not.

2. Special Purpose (Scientific Reference) Zones afford a high level of protection for marine flora and fauna. From vessels, rock lobster fishing is the only extractive activity permitted. From shore, line fishing, netting, rock lobster and abalone fishing are also permitted.

3. General Use Zones are those areas of the marine park not included in sanctuary, special use or recreational zones. All activities are permitted in general use zones provided they do not compromise the ecological values of the marine park.

Table 1. Areas of ten sanctuary zone locations in the Jurien Bay Marine Park.

Name	Area (hectares)
Fisherman Islands Sanctuary Zone	473
North Head Sanctuary Zone	204
Pumpkin Hollow Sanctuary Zone	99
Boullanger Island Sanctuary Zone	1,334
Booker Rocks Sanctuary Zone	7
Nambung Bay Sanctuary Zone	215
Cavanagh Reef Sanctuary Zone	261
Grey Sanctuary Zone	259
Target Rock Sanctuary Zone	198
Wedge Island Sanctuary Zone	11

Table 2. Areas of three special purpose (scientific reference) zone locations in the Jurien Bay Marine Park.

Name	Area (hectares)
Fisherman Islands Special Purpose (Scientific Reference) Zone	2,266
Hill River Special Purpose (Scientific Reference) Zone	4,190
Green Islands Special Purpose (Scientific Reference) Zone	7,582

In the present report, we summarise results of surveys of reef assemblages in the JBMP undertaken from mid October to early November in 1999, 2000, 2003, 2004, 2006 and 2007. The 2006 and 2007 surveys follow prohibitions on fishing, while the initial four surveys are regarded as representative of baseline conditions. Results of surveys from 1999 to 2004 have been outlined in previous reports (Barrett *et al.* 2002; Edgar *et al.* 2003; 2005).

The survey methodology focuses on reefs because these ecosystems are currently the most heavily exploited in the region and the most likely to show change following protection. Surveys were designed to provide as much quantitative information on assemblages of fishes, invertebrates and macroalgae within the limited dive time available. This methodology allows detection of predictable changes in heavily-exploited species, and also any unpredictable cascading ecosystem effects of fishing, as well as long-term patterns of regional change.

2. Methods

2.1 Sites

Underwater visual censuses of fishes, large mobile invertebrates and macroalgae were undertaken from 1999 to 2003 at a total of 25 sites distributed across the major management zone types (general use, sanctuary and scientific reference area). An additional 17 sites were added to the monitoring program in 2004, 2006 and 2007, making 42 sites in total (Fig. 1). Sites examined extended from moderately sheltered reefs at 2 m depth to reefs exposed to oceanic swell at 12 m depth.

Site locations were selected to provide a balance between the different management zone types, and also inshore and offshore reefs, with the constraint that they needed to be of sufficient size for placement of a 200 m length transect. Fourteen sites were surveyed in each of the general use, scientific reference and sanctuary zone types.

The position of each site was recorded using a hand held GPS (Scoutmaster) based on the WGS84 Datum System, with position recorded in degrees and decimal minutes. Position was post processed in 1999 to overcome difficulties associated with selective availability of the GPS system. This was not necessary in subsequent years. Site positions and site details are listed in Table 1. All data were entered onto an Excel spreadsheet.

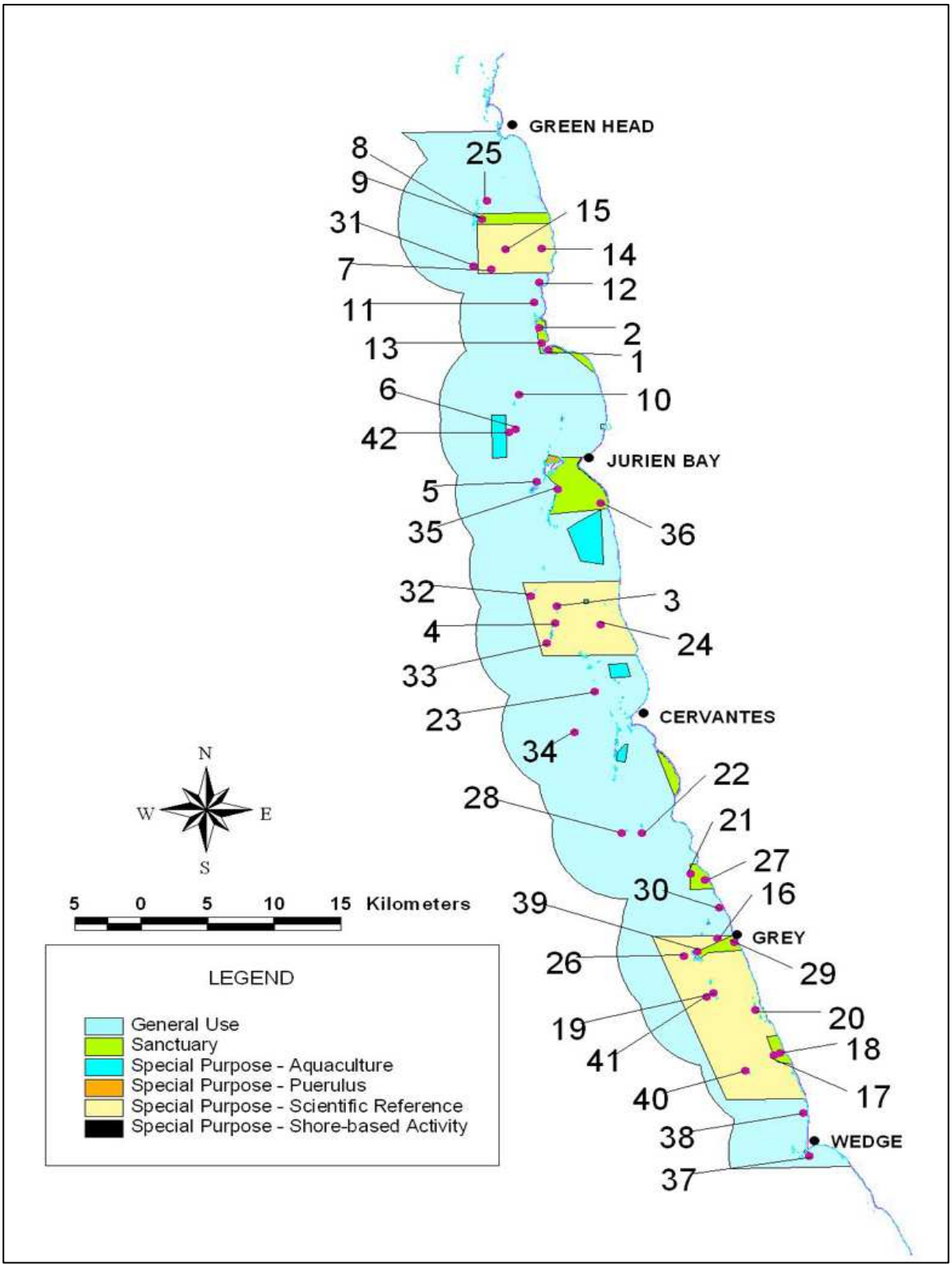

Figure 1. Map showing the location of sites surveyed within the Jurien Bay MPA.

2.2 Census methodology

At each reef site the abundance and size structure of large fishes, the abundance of cryptic fishes and benthic invertebrates, and the percent cover of macroalgae, corals and other cover-forming invertebrates, were each censused separately along four 50 m long transects (Edgar & Barrett, 1999; Barrett & Buxton, 2002). The transect lines were laid end to end along a fixed depth contour. For reefs that were relatively flat with no obvious contour to follow, sketch maps were created to allow similar positions to be relocated on subsequent surveys.

For fish transects, the density and estimated size-class of fish within 5 m of each side of the line were recorded on waterproof paper, with the diver swimming up the offshore side of the line and then back along the inshore side in the middle of a 5 m wide lane. Size-classes of total fish length used in the study were 25, 50, 75, 100, 125, 150, 200, 250, 300, 350, 375, 400, 500, 625, 750, 875 and 1000+ mm. Lengths of fish >1 m length were individually estimated.

Double counting of individual fish sometimes occurred when the diver returned along the inshore side of the transect line. Nevertheless, such double counts have little importance if the inshore and offshore 50 m x 5 m blocks are considered as two separate (albeit non-independent) estimates for the 50 m transect length. The reason that fish were counted on the return leg regardless of whether they were recognised as having been counted on the initial leg was that if this had not been done then return counts would be lower than initial counts, and mean total density estimates not comparable with 50 m x 5 m density estimates of workers elsewhere. Return counts were undertaken to allow greater precision of site estimates with little extra underwater time—transects already having been set.

Fish census data clearly are affected by a range of biases, including observer error and variation in behavioural responses of fish to divers (DeMartini and Roberts 1982; Kulbicki and Sarramega 1999; Thompson and Mapstone 1997). Such biases were investigated in part and discussed for the transect methods used here by Edgar *et al*. (2004). Despite the existence of census biases, we consider them to be largely systematic and not greatly confound interpretation of patterns because data are used for relative comparisons between different management zones. Care was taken to ensure that sampling effort for each diver was equitably distributed between the different management zone types.

Cryptic fishes and megafaunal invertebrates (large molluscs, echinoderms, crustaceans) were counted along the transect lines used for the fish survey by recording animals within 1 m of one side of the line (a total of four 1 m x 50 m transects). In order to increase precision of estimates for rock lobsters, a species of particular interest, counts for this species were recorded along both sides of the transect line (ie, a total of eight 1 m x 50 m transects per site) from 2004.

The area covered by different macroalgal, coral, sponge and other attached invertebrate species was quantified by placing a 0.25 m^2 quadrat at 10 m intervals along the transect line and assessing the percent cover of the various plant species. Cover was determined by counting the number of times each species occurred directly under the 50 positions on the quadrat at which perpendicularly placed wires crossed each other (a total of 1.25

m^2 for each of the 50 m sections of transect line). Sessile organisms were not surveyed in 2007 and, because of limited dive bottom time at depth, were also not censused at five sites (numbered 31, 33, 34, 40 and 41; see Fig. 1 and Table 3) in 2004.

2.3 Data transformation and statistical analyses

Fish biomass estimates

Fish abundance counts and size estimates obtained during underwater surveys were converted to biomass estimates using length-weight relationships presented for each species (in some cases genus and family) in Fishbase (http://www.fishbase.org). In cases where length-weight relationships were described in Fishbase in terms of standard length or fork length rather than total length (as recorded by divers), additional equations provided in Fishbase allowed conversion between different length metrics. For improved accuracy in biomass assessments, the bias in divers' perception of fish size underwater was additionally corrected using regression relationships presented in Edgar *et al*. (2004).

In using these transformations, we recognise that estimates of fish abundance made by divers can be greatly affected by fish behaviour for many species (Edgar *et al.* 2004); consequently biomass determinations, like abundance estimates, can reliably be compared only in a relative sense (i.e. for comparisons with data collected using the same methods) rather than providing an accurate absolute estimate of fish biomass for a patch of reef.

Fish species were subdivided into four trophic categories – herbivore, planktivore, benthic carnivore and higher carnivore using information on diet provided in Fishbase (http://www.fishbase.org/search.php). Benthic carnivores were distinguished from higher carnivores on the basis of whether their diet predominantly consisted of molluscs, amphipods, isopods and polychaetes rather than other fishes, squid and decapods.

Multivariate analyses

Similarity in community structure between sites was analysed using Principal Coordinates Analysis (PCA). Data for all three raw data sets (fishes, macroinvertebrates and sessile biota) were initially combined into a total site-by-mean abundance matrix after square root transformation of data. Counts for different surveys within each site were averaged to provide a single number for each site-by-species combination. Species recorded from less than three sites were excluded.

In order to avoid domination of the combined taxa analysis by one of the three taxonomic groups, the fish, invertebrate and sessile data sets were combined after data in each had been standardized to a similar range in abundance. This was done by multiplying each value in the macro-invertebrate data set by the maximum value of any fish species recorded at a site and dividing by the maximum value of any invertebrate species recorded at a site. The same process was used to standardise the sessile biota data set to the same maximum value as for fishes and macroinvertebrates.

Environmental influences on community structure were identified by relating six major environmental variates (1. latitude, 2. distance offshore (km), 3. distance to general use zone boundary (km), 4. transect depth (m), 5. underwater visibility (m), and 6. wave exposure) to each of the first two principal coordinates using Pearson correlation. Site depth was recorded from SCUBA gauges. Underwater visibility was estimated as the maximum distance sighted by divers along transect lines. Wave exposure was estimated at each site using a four point scale: 1: sheltered conditions in lagoonal and other protected environments with little oceanic swell but wind waves; 2: sheltered coast open to limited swell; 3: coast open to moderate swell; and 4: coast open to full oceanic swell.

Associations of common species with major community types were also assessed by calculating correlations with PCA axes using square-root transformed species abundance data for each site and species.

Patterns of similarity were assessed using cluster analysis, where a Bray-Curtis similarity matrix calculated from the square-root transformed data set used for the PCA was clustered using average linkage. Sites showing >50% and >60% similarity to each other on the dendrogram ouput were encircled by an ellipse on the PCA, as plotted by the PRIMER program (Carr 1996).

In addition to PCA, which is a form of metric multidimensional scaling, data were also analysed using non-metric Multi-Dimensional Scaling (MDS) to produce the best graphical depictions of biotic similarities between sites. PCA and MDS were both run with the PRIMER program. For MDS, the data matrix showing mean abundance of species at each site was square root-transformed to reduce the influence of the most abundant species, and converted to a symmetric matrix of biotic similarity between pairs of sites using the Bray-Curtis similarity index, which is relatively insensitive to data sets with many zero values (Faith *et al.* 1987). Fish biomass data were double square root-transformed. These procedures follow the recommendations of Faith *et al* (1987) and Clarke (1993) for data matrices with numerous zero records.

Data input to matrices for MDS comprised mean values for all sites within each management zone before and after prohibitions on fishing (i.e. 1999-2004 and 2006-07). The usefulness of the two dimensional MDS display of biotic relationships is indicated by the stress statistic, which signifies a good depiction of relationships when <0.1 and poor depiction when >0.2 (Clarke 1993).

Analysis of Variance (ANOVA)

Based on a replicated Before-After-Control-Impact (BACI) monitoring design framework (Green 1979), data were analysed using ANOVA. The statistical view taken of the JBMP was that it comprised a seascape subdivided into a mosaic of a fixed number of fished and unfished zone locations, all of which were surveyed if appropriate monitoring habitat was present (i.e. reef extending for at least 200 m and with a width of at least 10 m for fish counts). Monitoring sites were randomly located within these zone locations.

We also considered but rejected an alternate statistical view of the JBMP where sites are nested within zone locations that are nested within the three major management zone

types. Models based on this scenario were not applied because zone location was fixed rather than random, as would be required in the case of nesting of factors. Zone location was considered fixed because all sanctuary and scientific reference locations with appropriate reef habitat within the JBMP were surveyed as part of this study.

In our model, three fixed factors and one interaction term were included in the ANOVA—'year ', 'zone', 'year x zone' and 'location'—where zone 'location' provides a blocking factor. The factor 'year' represents two time periods before and after restrictions on fishing 1999-2004 and 2006-07, respectively. Data for each site were averaged across survey years within each time period. This averaging process allowed the full set of 42 sites to be included in analyses even though not all sites were monitored in all years.

The factor 'zone' also included two levels: fished and unfished. Unfished sites included all sites within both sanctuary and scientific reference zones for population variates examined other than for those relating to rock lobsters. Rock lobster density and mean size were analysed with only sanctuary zones considered as unfished zones, given that fishing for this species was not prohibited in scientific reference zones. By combining unfished sites in scientific reference zones with those in sanctuary zones, more powerful tests could be applied than with a three zone (sanctuary, scientific reference and general use) analysis.

The factor 'location' included seven levels that corresponded with the major unfished MPA zones (Boullanger Island, Cavanagh , Fisherman Islands, Green Islands, Hill River; North Head, Wedge Island) plus nearby reference sites in the general use zone. Sites in the Target Rock and Grey Sanctuary Zones were grouped with sites in the surrounding Green Islands Scientific Reference Zone as one location, except for the rock lobster analyses where they represented unfished locations and Green Islands Scientific Reference Zone a fished location. Location was included in the ANOVA as a blocking factor (i.e., without interaction terms) to remove regional variation and thereby reduce the residual error term, allowing more powerful MPA-related tests.

The interaction term 'year x location' was the most important factor with respect to the identification of MPA effects. Any major recovery of fished populations within MPA zones should be indicated by a significant change in this factor, as it would indicate change in unfished zones relative to fished zones between the periods pre- and post-restrictions on fishing.

Following graphical assessment of the distribution of residuals, abundance and density data were log transformed to reduce heteroscedasticity in ANOVAs. Species richness data possessed homogeneous variance structure and were not transformed.

Correlation analyses

Much information on variation within and between zones is lost with an ANOVA approach because sites in all zones of the same type are considered equal. Factors that vary between sites and can affect a species' response to protection from fishing, such as distance from the reserve boundary, size of protected zone, or level of pre-existing fishing pressure, possesses intrinsic interest and are best recognised; however, the

ANOVA approach overlooks any such relationships, thereby adding to spatial noise between replicates.

Two spatial variates, distance from protected zone boundary and size of protected zone, have been found in other studies to greatly affect ecological responses to protection from fishing (Buxton *et al.* 2005; Côté *et al.* 2001). The significance of relationships between these variates and ecological change that follows protection from fishing were here assessed using Spearman rank correlations. For each metric examined, sites were firstly ranked in order of change, which was calculated as a mean value for each of the 42 sites surveyed in 2006-07 less the corresponding mean value for the period 1999-2004. Correlation coefficients were calculated using these rankings versus ranking of sites by distance from protected zone boundary and size of protected zone (Table 3).

Two-tailed tests were applied for the majority of metrics examined, given that the cascading ecosystem effects were possible and the direction of change consequently unpredictable. Nevertheless, metrics related to exploited species (large fishes, higher trophic level carnivorous fishes, dhufish, breaksea cod, baldchin groper, silver trevally, and rock lobsters) were examined using one-tailed tests because population numbers, biomass and mean size of species in these groups were predicted to increase.

Table 3. Site details for locations surveyed in Jurien Bay, including name of MPA zone location (or name of adjacent protected zone location if general use zone), depth, distance from protected zone boundary, area of sanctuary zone, area of protected zone (= sanctuary zone plus adjoining scientific reference zone), underwater visibility (Vis) and wave exposure index (Exp).

Site No.	Site name	Latitude	Longitude	Zone	MPA location	Depth (m)	Distance (km)	Area 1 (ha)	Area 2 (ha)	Vis (m)	Exp
1	North Head 1	30º13.912'	114º59.924'	Sanctuary	North Head	2	0.21			8.2	1
2	Sandland Island	30º12.914'	114º59.524'	Sanctuary	North Head	5	0.17			6.7	2
3	Outer Rocks-Inner Coffins	30º25.285'	115º0.116'	Scientific	Hill River	5	1.90	0	4190	11.2	3
4	Outer Rocks (north) 2	30º26.026'	114º59.984'	Scientific	Hill River	5	1.41	0	4190	9.7	3
5	Escape Island	30º19.745'	114º59.263'	General	Boullanger	5	-0.83	0	0	8.4	3
6	Inner Seaward Ledge	30º17.404'	114º58.349'	General	Hill River	5	-3.84	0	0	10.2	2
7	Juddy Reef	30º10.275'	114º57.33'	Scientific	Fishermans	5	0.19	0	2739	10.7	3
8	Fishermans Is 1	30º8.042'	114º56.935'	Sanctuary	Fishermans	3	0.30	473	2739	12.3	1
9	Fishermans Is 2	30º8.042'	114º56.935'	Sanctuary	Fishermans	3	0.30	474	2740	11.3	1
10	North Tail	30º15.87'	114º58.5'	General	Hill River	6	-3.26	0	0	13.5	3
11	Australia Lump	30º11.788'	114º59.316'	General	North Head	4	-1.21	0	0	7.0	3
12	Sandy Cape	30º10.882'	114º59.577'	General	North Head	2	-0.46	0	0	9	2
13	North Head Island	30º13.61'	114º59.611'	Sanctuary	North Head	4	0.17	204	204	8.9	3
14	North Lumps	30º9.412'	114º59.73'	Scientific	Fishermans	2	1.70	0	2739	10.9	2
15	Middle Lumps	30º9.407'	114º58.011'	Scientific	Fishermans	5	1.69	0	2739	11.3	3
16	Longman Reef (off Grey)	30º40.131'	115º7.316'	Scientific	Green	3	0.19	0	7582	9.9	2
17	Flat Rock	30º45.343'	115º9.898'	Sanctuary	Target Rock	3	0.03	198	8039	13.6	2
18	Flat Rock Reef	30º45.249'	115º10.174'	Sanctuary	Target Rock	4	0.27	199	8039	8.9	2
19	Gazely Reef	30º42.557'	115º7.084'	Scientific	Green	4	2.37	0	8039	12.4	3
20	Kearn Reef	30º43.322'	115º9.042'	Scientific	Green	4	4.73	0	8039	12.6	2
21	Cavanagh Reef	30º37.246'	115º6.143'	Sanctuary	Cavanagh	5	0.03	261	261	9.0	2
22	Inner Seven Ft Reef	30º35.397'	115º3.889'	General	Green	4	-4.20	0	0	13.5	3
23	Sams Reef	30º29.108'	115º1.799'	General	Cavanagh	5	-2.71	0	0	12.4	3
24	No Name Reef	30º26.111'	115º2.13'	Scientific	Hill River	3	2.20	0	4190	7.4	2
25	Fishermans Island	30º7.244'	114º57.219'	General	Fishermans	4	-0.79	0	0	12.4	2
26	Outer Green Islands	30º40.886'	115º5.729'	Scientific	Green	12	1.43	0	8039	13	4
27	Cavanagh Reef	30º37.53'	115º6.802'	Sanctuary	Cavanagh	3	0.51	261	261	6.2	2
28	Outer Seven Foot Rocks	30º35.391'	115º2.923'	General	Green	10	-5.56	0	0	12.6	4
29	Inshore Grey	30º40.3'	115º8.138'	Sanctuary	Grey	5	0.40	259	8039	5.0	2
30	Inshore Grey North	30º38.772'	115º7.434'	General	Cavanagh	3	-1.27	0	0	6.8	2
31	Main Reef	30º10.15'	114º56.53'	General	Fishermans	10	-0.21	0	0	9.3	4
32	Offshore Hill River	30º24.8'	114º58.9'	Scientific	Hill River	10	0.21	0	4190	12.7	4
33	Offshore Outer Rocks	30º26.908'	114º59.579'	Scientific	Hill River	10	0.41	0	4190	11.9	4
34	Big Wave Reef	30º30.87'	115º0.82'	General	Hill River	10	-5.56	0	0	11.9	4
35	Midshore Boullanger Is	30º20.08'	115º0.24'	Sanctuary	Boullanger	2	0.03	1334	1334	10.0	1
36	Inshore Boullanger Is	30º20.748'	115º2.2541'	Sanctuary	Boullanger	5	0.34	1334	1334	6.0	1
37	Wedge Island	30º49.868'	115º11.463'	Sanctuary	Wedge	5	0.03	11	11	6.0	2
38	North Wedge	30º47.951'	115º11.217'	General	Wedge	5	-1.06	0	0	6.1	2
39	SE Green Is	30º40.693'	115º6.36'	Sanctuary	Grey	3	0.03	259	8039	6.2	1
40	Offshore Target Rocks	30º46.005'	115º8.518'	Scientific	Green	10	2.19	0	8039	8.8	3
41	Offshore Gazaly Reef	30º42.716'	115º6.763'	Scientific	Green	10	1.76	0	8039	7.9	4
42	Outer Seaward Ledge	30°17.426’	114°57.990’	General	Hill River	10	-3.84	0	0	14.1	4

3. Results and discussion

3.1 Biotic similarities between sites

The Principal Component Analysis (PCA) was initially run using mean data for each of the 42 sites for periods before (1999-2004) and after (2006-07) fishing restrictions. The first canonical axes was very highly correlated with distance offshore (r = -0.68), and also highly correlated (r > 0.5) with transect depth, underwater visibility and wave exposure (Table 4). These three variates were also highly correlated with each other, as offshore sites tended to have high levels of wave exposure, clear water, and were censused at 10 m depth rather than 5 m or 2 m because of excessive wave surge in the shallows.

The second canonical axis was highly correlated (r = 0.45) with latitude, while the third axis was most highly correlated with wave exposure (r = 0.44) and depth (r = 0.42). The community data set did not appear to be much affected by MPA effects, as indicated by the variate "distance to general use zone boundary" lacking any clear correlation with a PCA axis. This variate was defined as distance from boundary inside sanctuary zones and scientific reference zones for 2006-07 data, and was given a value of 0 for data for sites surveyed before 2005 and also sites surveyed in general use zones after 2005.

Table 4. Pearson correlation coefficients between environmental variates and first three PCA axes, using 42 sites and mean data for two survey periods (before MPA and after MPA).

PCA axis	Latitude	Distance offshore	Distance to MPA zone boundary	Depth	Visibility	Wave exposure
PCA1	-0.22	-0.68	-0.05	-0.57	-0.55	-0.64
PCA2	0.45	0.36	-0.06	-0.08	0.15	-0.28
PCA3	0.20	-0.14	0.08	-0.42	-0.06	-0.44

Outcomes of the PCA are summarised in Fig. 2, where, because MPA effects were negligible, mean data for each site across all surveys are plotted against the first two canonical axes. Sites located on exposed offshore reefs tended to group to the left of this figure, while northern sites tended to group towards the top of the figure, although the single site surveyed in the southern Wedge Island sanctuary zone (#37) also groups at the top with sheltered Fisherman Islands sites. Neither scientific reference zone nor general reference zone sites show much biotic overlap with sanctuary zone sites, which tended to be concentrated towards the right of the figure.

Correlation coefficients between environmental variates and PCA axes for the data set used to generate Fig. 2 were slightly higher than those shown in Table 4 because data for each site were averaged over the full survey period, and thus involved more precise estimates of site means. This averaging was possible because time (i.e. MPA) effects were so slight.

Distance offshore was again extremely highly correlated with the first three PCA axes (r = -0.70, 0.39, and 0.12 for PCA1, 2, and 3, respectively), with a combined r-value of 0.81, which is equivalent to 66% of total variance explained. Wave exposure was even more highly correlated with the first three PCA axes (r = -0.66, -0.18, and 0.54 for PCA1, 2, and 3, respectively), with a combined r-value of 0.87 (= 76% of total variance).

The species with abundances most highly correlated with the first two PCA axes were the kelp *Ecklonia radiata* (combined r = 0.63), the western buffalo bream *Kyphosus cornelii* (combined r = 0.46), and the sea urchin *Heliocidaris erythrogramma* (combined r = 0.40). Each of these species also characterised a particular grouping of sites and species (Fig. 2). *Ecklonia radiata* typified wave exposed sites and generally co-occurred with the red seaweeds *Pterocladia lucida* and *Hennedya crispa*. *Kyphosus cornelii* was primarily associated with sites in the vicinity of Fisherman Islands, while *H. erythrogramma* was most commonly found on inshore southern reefs in association with the red seaweeds *Dictymenia sonderi* and *Neurymenia fraxinifolia*.

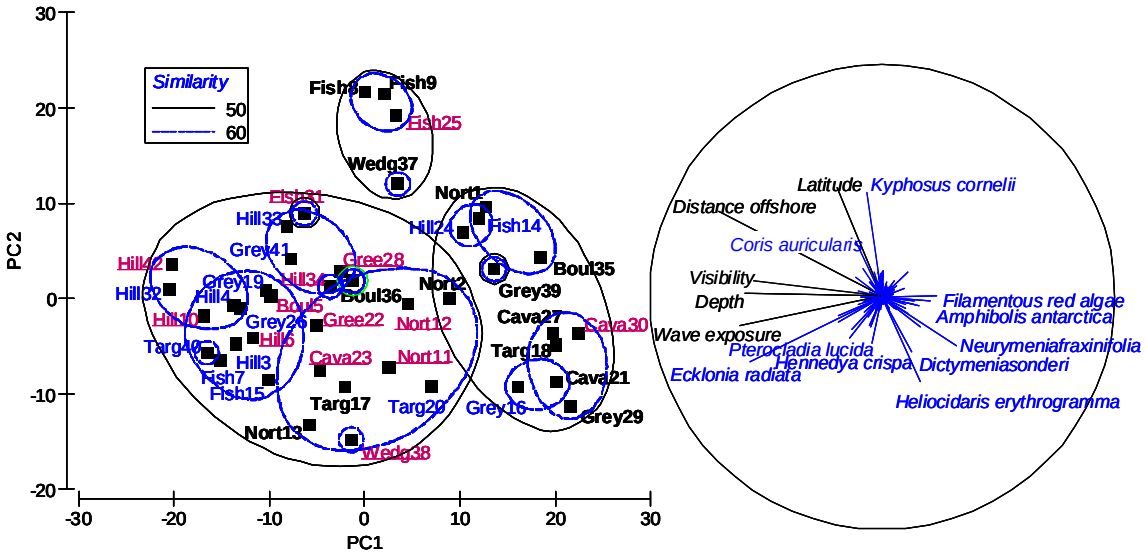

Figure 2. Results of PCA showing biotic relationships between sites. Each site is depicted using the first four letters of the zone (Boul: Boullanger Island; Cava: Cavanagh ; Fish: Fisherman Islands; Gree: Green Islands; Hill: Hill River; Nort: North Head; Targ: Target Rock; Wedg: Wedge Island; Table 4) followed by site number as listed in Table 3. Site codes are underlined and in red for general use zones, in blue for scientific reference zones, and in bold for sanctuary zones. Contour plots around groups encompass sites that group at either >50% and >60% similarity in cluster analysis. Correlation coefficients relating first two PCA axes and environmental variates are also shown (black), as are correlations relating PCA axes and mean square root-transformed abundance for species showing strongest relationships (blue). Circle radius indicates r = 1.

Community-level changes in plant and animal abundances in different management zones are depicted using MDS in Fig. 3 for the periods before (1999-2004) and after prohibitions on fishing (2006-07). In this analysis, all three major taxonomic groups were aggregated using the same site-abundance matrix as used for the PCA, and mean values for sites in sanctuary zones and in scientific reference sites calculated, and input to form the Bray-Curtis similarity matrix. Thus, this plot includes data averaged across multiple sites within each zone. Zones with high levels of biotic similarity lie adjacent to each other, while sites with few similarities are positioned at distance. A moderate stress level is associated with this figure (0.17), indicating that it provides a reasonable but not great two-dimensional depiction of relationships.

Sanctuary zone locations tended to be outliers located to the right and top of Fig. 3, while the three scientific reference zone locations were central and largely overlapped. Thus, sites studied in scientific reference zones possessed similar biotas to each other and were also similar to reference sites in general use zones, whereas reef communities at sites in sanctuary zones showed many differences across the JBMP region with few good comparative reference sites.

Reef communities in scientific reference zone locations changed very little following prohibitions on fishing (Fig. 3). By contrast, biotas in sanctuary zone locations exhibited substantial apparent change between time periods. This difference presumably relates in part to the greater number of sites sampled within each scientific reference zone location, as this would act to reduce the variability associated with differences between real and estimated means, and consequently also differences between estimated means in different sampling periods.

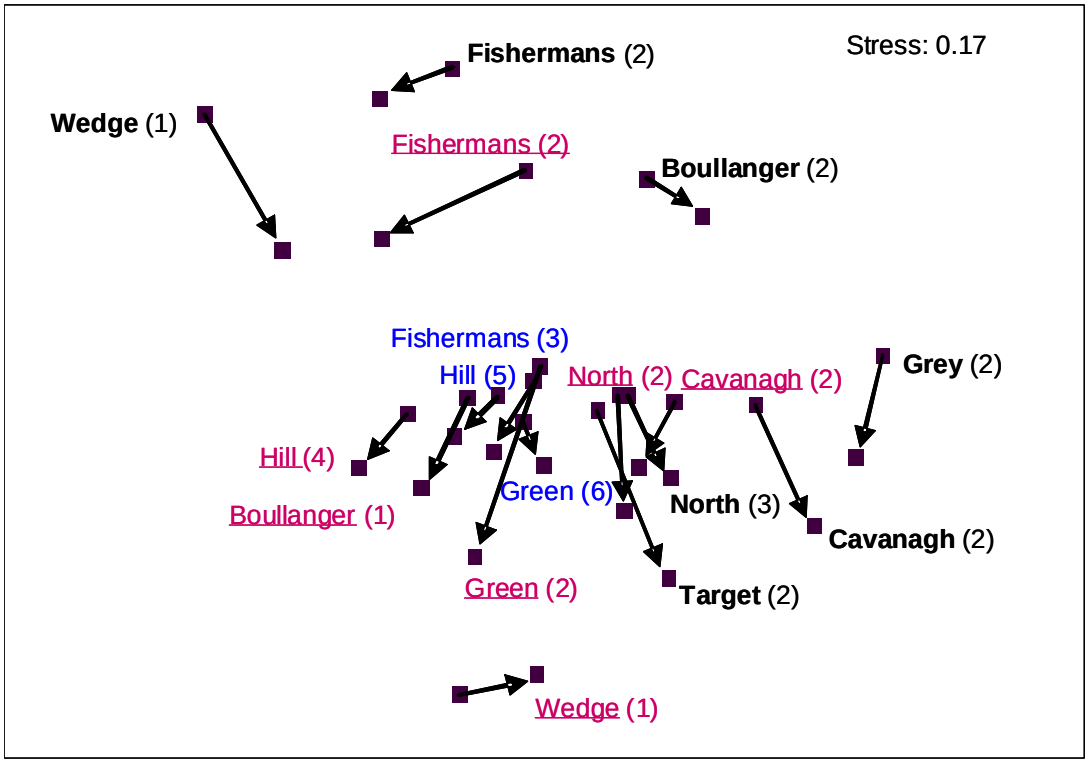

Figure 3. MDS plot showing relationships between zone locations for total community abundance before and after MPA declaration. Changes at different zones in mean fish abundance from the period 1999-2004 to the period 2006-07 are indicated by arrows. Site codes are underlined and in red for general use zones, in blue for scientific reference zones, and in black bold for sanctuary zones, with number of sites surveyed in each zone indicated in parentheses.

Given that the MDS plot illustrated greater change in sanctuary zone locations following prohibitions on fishing than for scientific reference zones and general use zones, and that these results may have been confounded by a lesser number of sites monitored in each sanctuary zone location, data were also analysed using means calculated for the whole region rather than for each location. Sites in general use zones were subdivided into sets of inshore (<1 km from coast) and offshore (>1 km from coast) sites, with inshore sites used as reference sites for the sanctuary zones and offshore sites used as reference sites for general use zones. This was possible because distance offshore was found in the PCA to explain much of the biotic variation between sites, and sites in sanctuary zones and scientific reference zones were concentrated in the inshore and offshore regions, respectively.

Results of MDS using this dataset indicated that change in scientific reference zones following prohibition on fishing was of similar magnitude and direction to change in offshore sites in general use zones (i.e. SRZ reference in Fig. 4). Change in sanctuary zones was of similar direction but lesser magnitude than change in inshore general use zones, and the trajectories of change for these two zone types overlapped.

The large apparent change amongst inshore sites in general use zones presumably resulted from only four sites in this grouping having been surveyed, compared to 10-14 sites surveyed within each of the other three groupings. Regardless, the actual level of change in inshore general use zones was relatively low, as evident in cluster analysis plots (>70% Bray-Curtis similarity; Fig. 4).

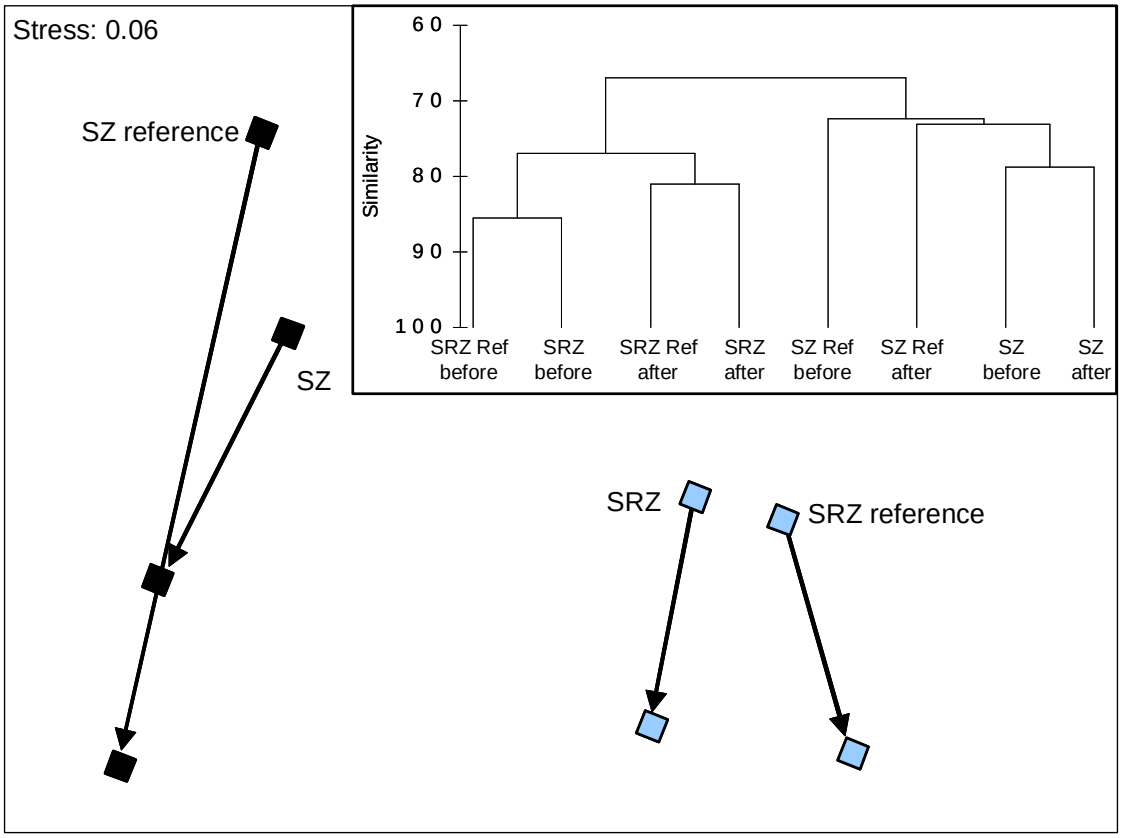

Figure 4. MDS plot showing relationships between zones for total biota before and after MPA declaration. Changes in density from the period 1999-2004 to the period 2006-07 in sanctuary zones (SZ), scientific reference zones (SRZ), inshore general use zones (SZ reference) and offshore general use zones (SRZ reference) are indicated by arrows using mean site values. Results of cluster analysis (average linkage) using the same similarity matrix are also shown.

Fish abundance and biomass

Mean fish abundance at sites within different MPA zones generally varied little from pre-MPA to post-MPA periods (Fig. 5), particularly for scientific reference zones. The largest changes were associated with the Cavanagh and North Head sanctuary zones, although the reference sites in general use zones associated with these sanctuary zones also showed relatively large changes, indicating that effects may be more closely related to regional changes than protection from fishing. Changes in sanctuary zones following protection of fishes tended to manifest on the Fig. 5 plot as upward movement of zones, whereas changes in general use and scientific reference had little consistent direction.

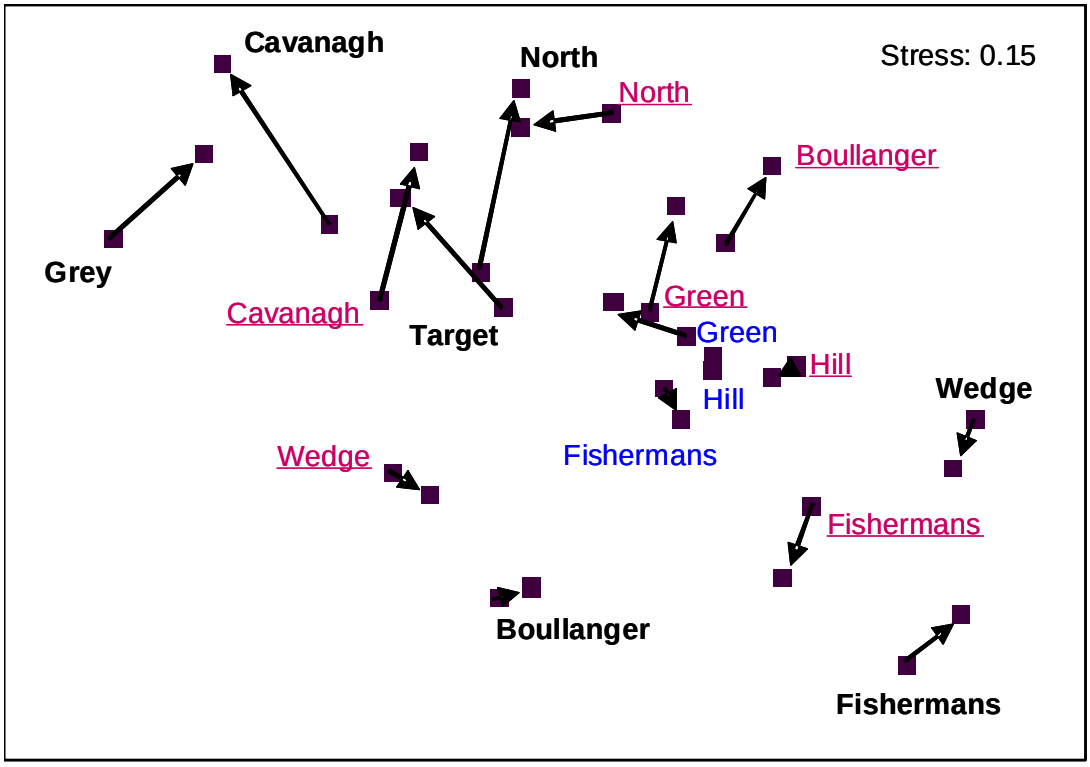

Figure 5. MDS plot showing relationships between zone locations for fish abundance before and after MPA declaration. Changes at different zones in mean fish abundance from the period 1999-2004 to the period 2006-07 are indicated by arrows. Site codes are underlined and in red for general use zones, in blue for scientific reference zones, and in black bold for sanctuary zones.

Patterns of biotic similarity between sites and survey periods for fish biomass were similar to those based on fish abundance (Fig. 6); however, the amount of change between survey periods was greater for fish biomass, presumably because errors associated with site means could be affected by an anomalous count of one large fish. Regardless, stress associated with this plot was again moderate (0.14), indicating a reasonable depiction of relationships. For both fish abundance and fish biomass plots, scientific reference areas tended to be biologically very similar to each other and be located near the centre of the figure, whereas sanctuary zones exhibited a large range of variation and occurred at the extremities of the figure. The greatest biological differences between sites, as indicated by separation in the MDS plot, were between the northern Fisherman Islands Sanctuary Zone and the southern Cavanagh and Grey Sanctuary Zones.

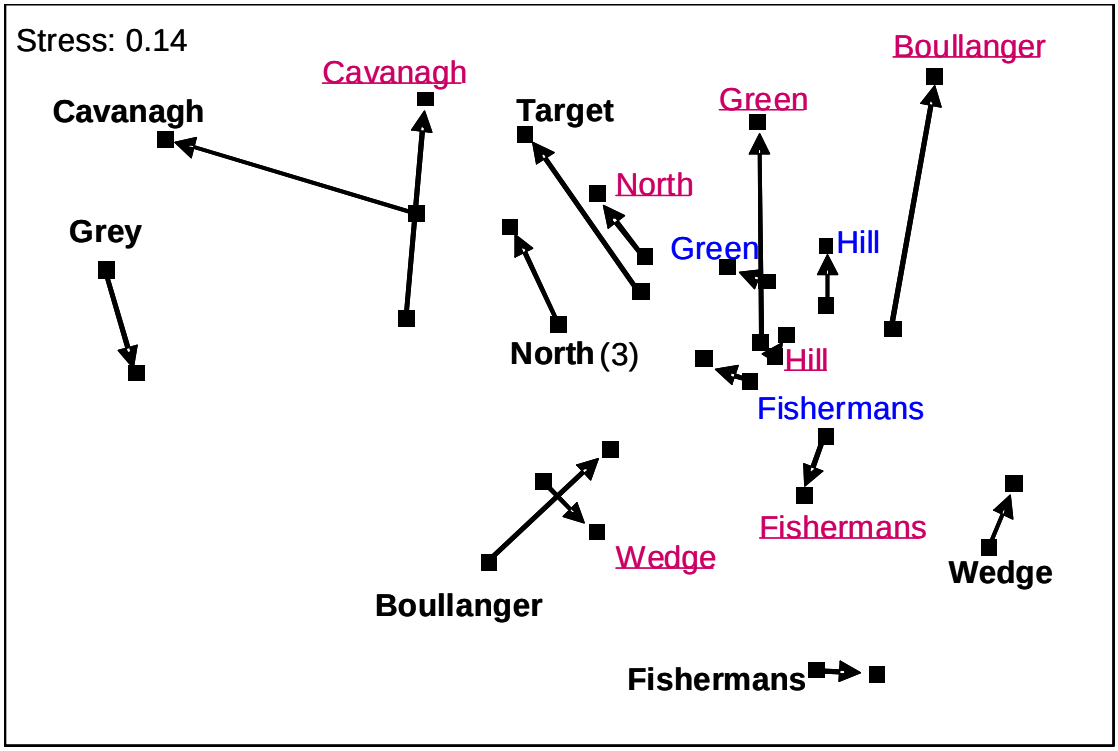

Figure 6. MDS plot showing relationships between zone locations for fish biomass before and after MPA declaration. Changes at different zones in mean fish biomass from the period 1999-2004 to the period 2006-07 are indicated by arrows. Site codes are underlined and in red for general use zones, in blue for scientific reference zones, and in black bold for sanctuary zones.

Macro-invertebrate abundance and sessile organism percent cover

The MDS plot for macro-invertebrate abundance exhibited many of the same patterns as for fishes, albeit with even greater variation between survey periods (Fig. 7). Care should be taken when interpreting this figure as the stress value was 0.20, indicating that much of the variance between sites cannot be accommodated in a two-dimensional plot. The Green Islands General Use Zone showed the most extreme change from pre- to post-MPA survey periods. This outcome was a consequence of very few invertebrates being recorded at the two sites in this zone (Outer Seven Foot Rocks), and a large associated stochastic variability between surveys.

As in other figures, scientific reference zones are central in the MDS plot for percentage cover of sessile organisms, and show little variation between survey periods (Fig. 8). Sanctuary zones all tend to be located to the right and bottom of the plot while general use zones tend fall towards the top left of the figure. Thus, interspersion of the three zone types is poor, with different assemblage types predominating in the different management zone types. The site in the general use zone adjacent to the Wedge Island Sanctuary Zone provides a poor reference location for the Wedge Island Sanctuary Zone site despite being located only 3.2 km distant; ecological communities at the reference location mapped onto the opposite side of the MDS figure when compared with the Wedge Island Sanctuary Zone site.

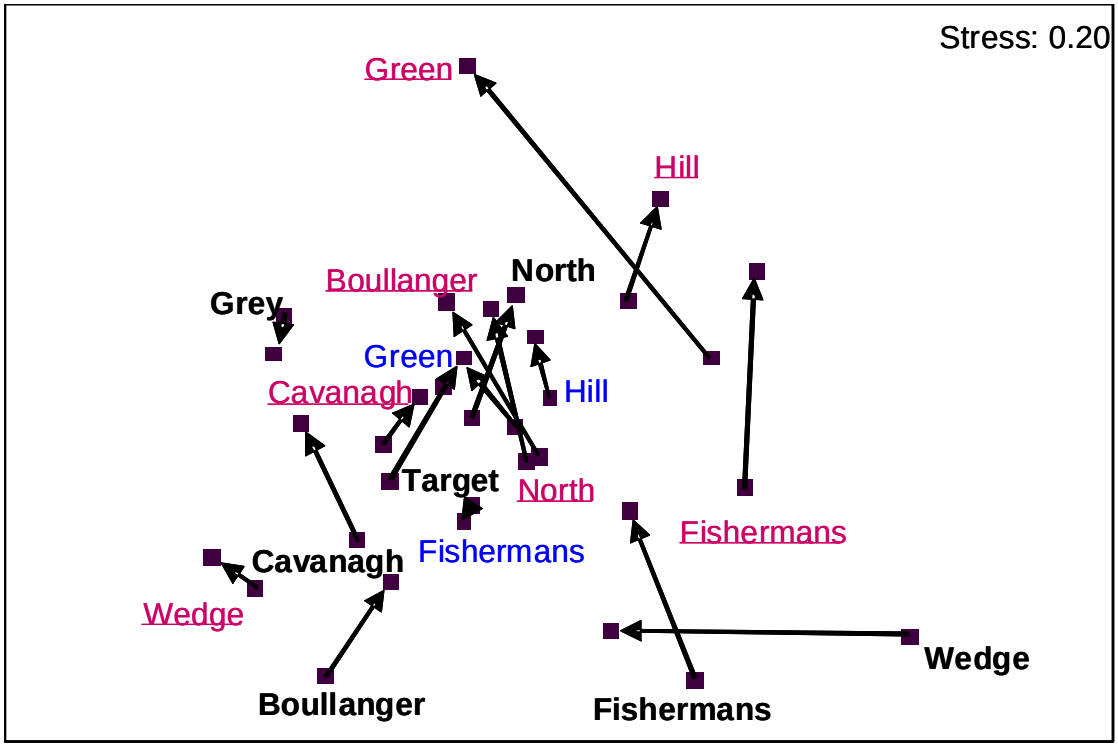

Figure 7. MDS plot showing relationships between zone locations for macro-invertebrate abundance before and after MPA declaration. Changes at different zones in mean fish biomass from the period 1999-2004 to the period 2006-07 are indicated by arrows. Site codes are underlined and in red for general use zones, in blue for scientific reference zones, and in black bold for sanctuary zones.

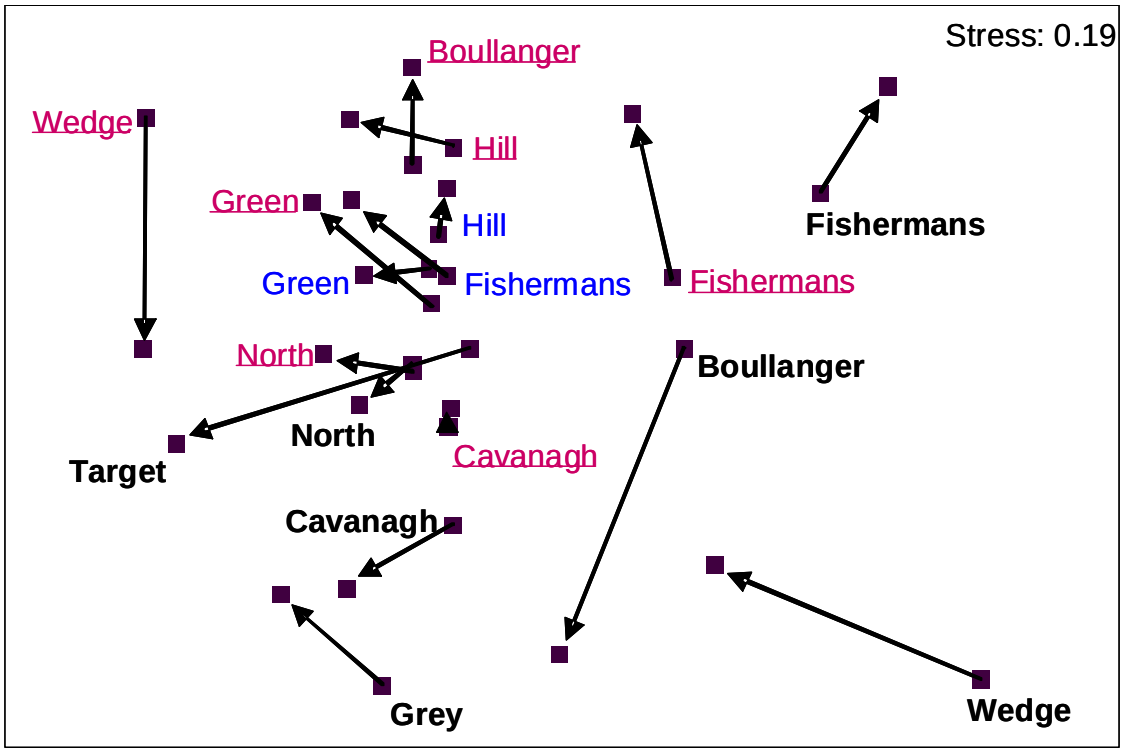

Figure 8. MDS plot showing relationships between zone locations for percent cover of sessile organisms (seaweeds, seagrasses, sessile invertebrate) before and after MPA declaration. Changes at different zones in mean fish biomass from the period 1999-2004 to the period 2006-07 are indicated by arrows. Site codes are underlined and in red for general use zones, in blue for scientific reference zones, and in black bold for sanctuary zones.

3.2 Patterns of species richness

Patterns of biodiversity at the scale of site have been assessed using total number of species recorded at a site during each survey period. Results were highly consistent between management zones and survey periods for total fishes and fishes in different trophic groups, with an average of ≈21 fish species sighted at each site (Figs. 9 and 10). No significant differences in species richness between zones or between years were evident when data were analysed using ANOVA (Table 5), other than for benthic carnivores, which showed a slight decline through time (Fig. 10).

Table 5. Mean squares (MS), F-value (F) and significance (*: 0.05>p>0.01; **: 0.01>p>0.001; ***: 0.001>p) resulting from ANOVAs using time and MPA effect as factors, and zone location as a blocking factor, for species richness data (number of species recorded per site). Degrees of freedom are 1 (time), 1 (MPA effect), 6 (zone location), 74 (error associated with fishes and macro-invertebrates).and 69 (error associated with fishes and macro-invertebrates).

Taxon	Time		MPA		Zone location		Time x MPA		Error
	MS	F	MS	F	MS	F	MS	F	MS
Herbivores	0.335	0.221	0.139	0.091	2.651	1.746	0.251	0.166	1.518
Planktivores	4.920	2.579	2.155	1.129	7.098	3.720**	0.018	0.010	1.908
Benthic carnivores	51.206	5.529*	15.811	1.707	20.349	2.197	9.643	1.041	9.261
Higher carnivores	2.149	1.172	1.381	0.753	7.229	3.942**	0.381	0.208	1.834
Large (>250 mm) fishes	7.085	0.992	2.303	0.322	18.741	2.623*	0.121	0.017	7.144
Total fishes	72.024	2.277	10.962	0.347	86.639	2.739*	3.429	0.108	31.628
Macro-invertebrates	1.524	0.163	0.388	0.042	24.610	2.638*	4.024	0.431	9.329
Red macroalgae	262.270	11.550***	77.000	3.391	48.302	2.127	23.184	1.021	22.708
Green macroalgae	1.176	0.456	0.047	0.018	6.48	2.509*	1.995	0.773	2.582
Brown macroalgae	1.616	0.361	6.127	1.371	7.328	1.639	0.048	0.011	4.470
Total macroalgae	191.519	5.360*	42.476	1.189	99.489	2.785*	36.099	1.010	35.728

Amongst the trophic groups, the number of fish species was highest for benthic carnivores followed by herbivores, and lowest for higher carnivores and planktivores, which were both poorly represented at all sites in all zones (Fig. 10). The latter two trophic groups were, however, non-randomly distributed across the JBMP, as indicated by a significant effect of zone location in ANOVAs (Table 5).

None of the fish species richness metrics examined exhibited a detectable change following protection from fishing. Neither the 'time x MPA' factor in the ANOVA (Table 5), nor correlations between change in species richness following restrictions on fishing and either protected zone size or distance from protected zone boundary (Table 4) generated a significant result.

The number of mobile macro-inverteibrate species at sites remained highly constant between years and in the three major zone types, with an average of 11.2 species observed per site (Fig. 11). Significant variation was, however, evident between different zone locations (Table 5).

Macroalgal species richness varied significantly between survey periods (Table 5), with lowest numbers of species recorded on the first two survey occasions (Fig. 12). This trend was driven by increasing numbers of red algal species, while numbers of brown and green

algal species showed no trend for change with time. The number of green macroalgae and total macroalgae varied between locations; however, red and brown macroalgae occurred in similar species numbers per site across all zone locations (Table 5).

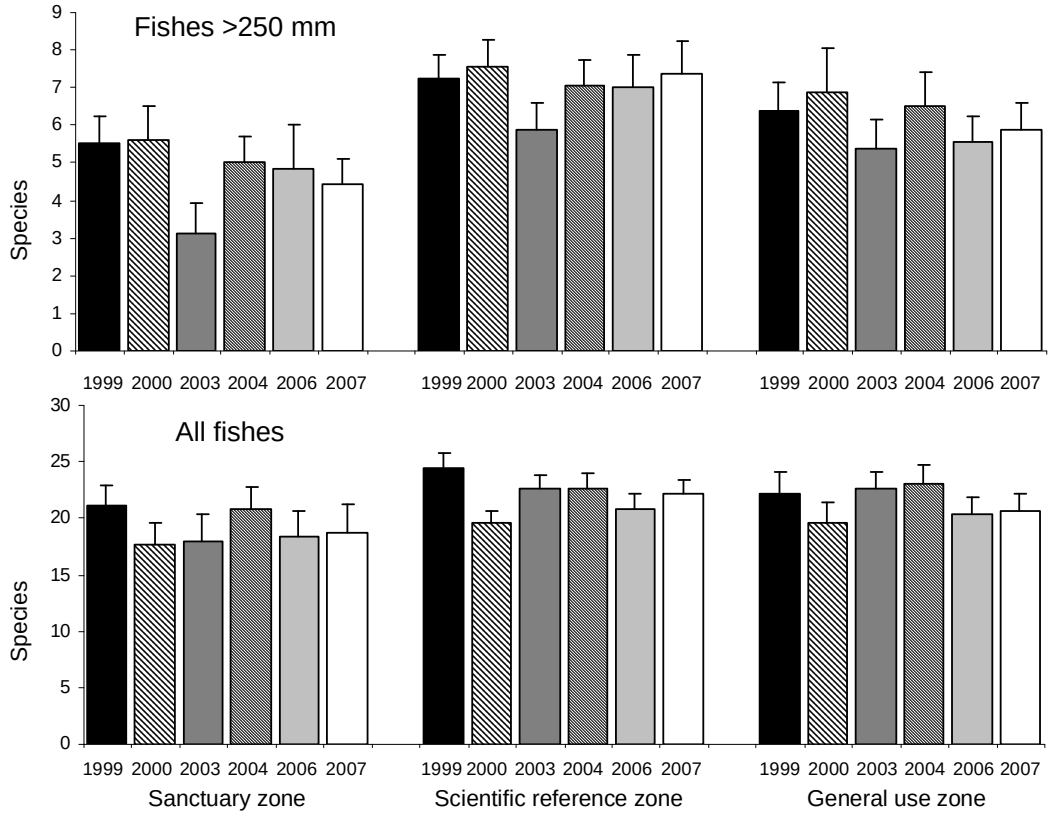

Figure 9. Mean species numbers of all fishes and large (>250 mm) fishes observed per site (± SE) in different management zones in different years.

Table 6. Spearman rank correlations relating change in species richness of major taxonomic groups following protection from fishing with distance of site from protected zone (= sanctuary zone + scientific reference zone) boundary and with size of protected zone. None of the correlations were significant at p = 0.05.

Taxon	Distance to protected area boundary	Area protected
Herbivores	0.04	0.12
Planktivores	-0.11	0.13
Benthic carnivores	0.06	0.05
Higher carnivores	-0.01	0.11
Large (>250 mm) fishes	-0.02	0.21
Total fishes	0.01	0.12
Macro-invertebrates	0.03	-0.22
Red macroalgae	0.13	0.11
Green macroalgae	0.16	-0.16
Brown macroalgae	0.12	0.09
Total macroalgae	0.15	0.06

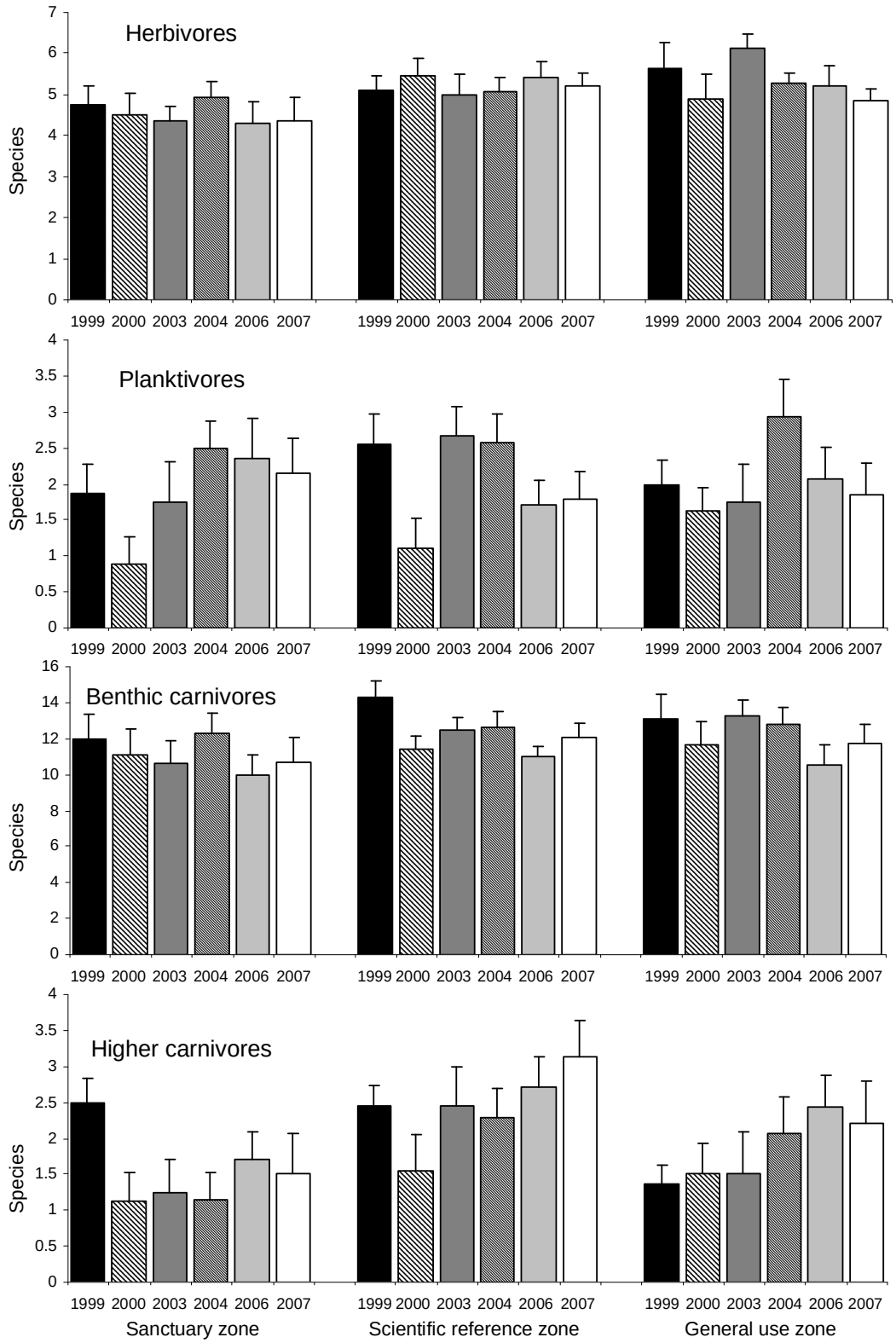

Figure 10. Mean number of fish species belonging to four major trophic groups observed per site (± SE) in different management zones in different years.

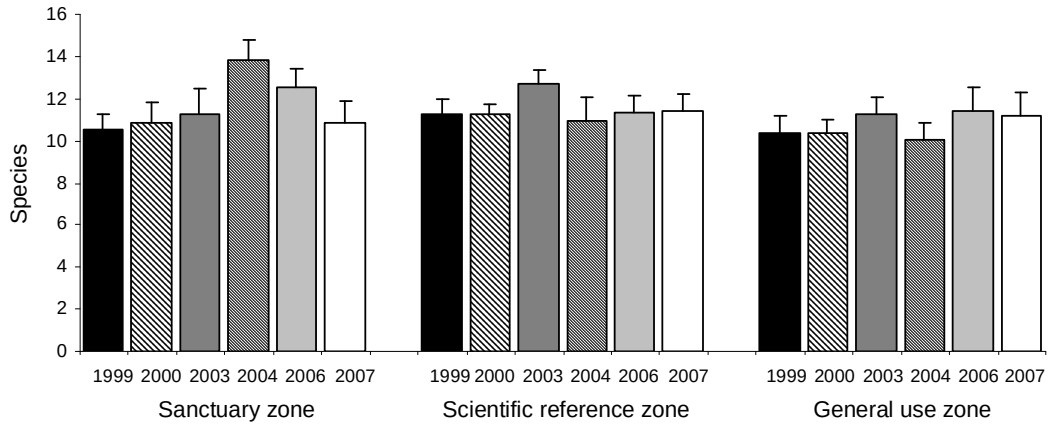

Figure 11. Mean number of mobile macro-invertebrate species observed per site (± SE) in different management zones in different years.

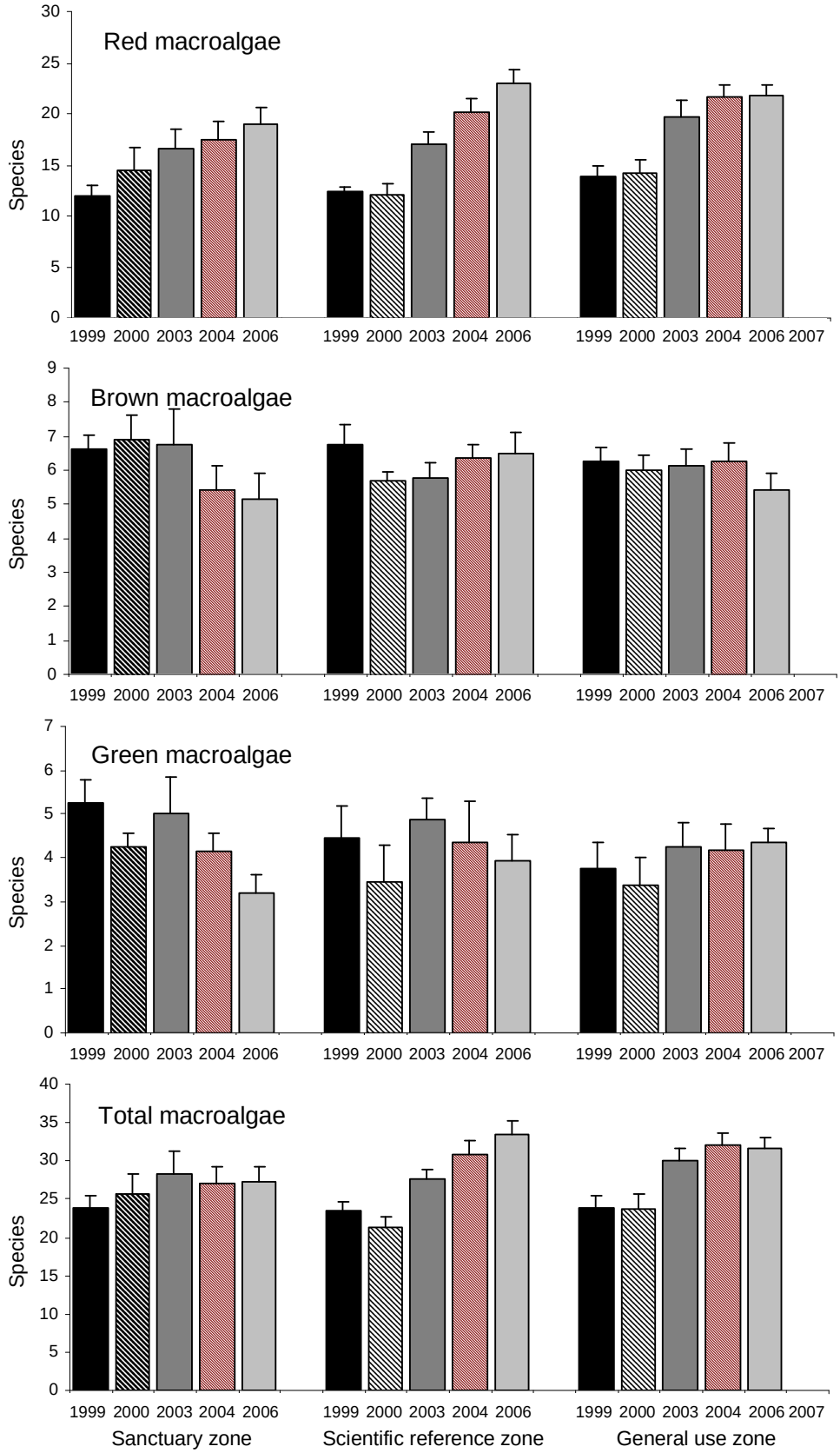

Figure 12. Mean number of red, brown, green and total macroalgal species observed per site (± SE) in different management zones in different years.

3.3 Patterns of faunal and floral density

Fish abundance

None of the four community metrics associated with total fish abundance at sites showed a significant temporal change between the periods before and after restrictions on fishing (Fig. 13; Table 7). However, abundance of the baldchin groper *Choeroden rubescens* showed a significant increase (Fig. 14), and abundance of the brown-spotted wrasse *Notolabrus parilus* (Fig. 15) a significant decrease, across the region when assessed using ANOVA (Table 5).

Although the graph of total abundance of fishes at sites appeared to be relatively stable across zones (Fig. 13), this metric was found using ANOVA after log transformation to differ significantly between protected (sanctuary and scientific reference) and general use zones (Table 7). Total fish abundance was influenced by numbers of the abundant western king wrasse *Coris auricularis*, which occurred in significantly lower numbers in protected zones than general use zones (Fig. 15), whereas the moon wrasse *Thalassoma lunare* showed the opposite pattern (Fig. 16). Variation between zone locations within the JBMP was significant for several of the metrics examined, including total total abundance of large exploited fishes, and abundances of *Notolabrus parilus* and the western buffalo bream *Kyphosus cornelii*.

The primary test of interest in the ANOVA, the interaction between time and MPA effect, was not significant for any metric examined (Table 7), indicating that any change in the fish community in protected zones relative to general use zones following restrictions on fishing was relatively slight. Nevertheless, the total number of large exploited fishes and abundance of the dhufish *Glausosoma hebraicum* both increased significantly following restrictions on fishing in the largest protected areas (Table 8). The total abundance of exploited fishes increased 110% at sites in the two years following protection in sanctuary and scientific reference zones (Fig. 13), compared to an overall increase of 43% decrease over the same period in general use zones. Dhufish numbers increased 13% in two years in protected zones compared to an overall 29% decrease over the same period in general use zones (Fig. 14).

The western scalyfin exhibited a significant negative correlation between change over time and distance from protected area boundary (Table 8).

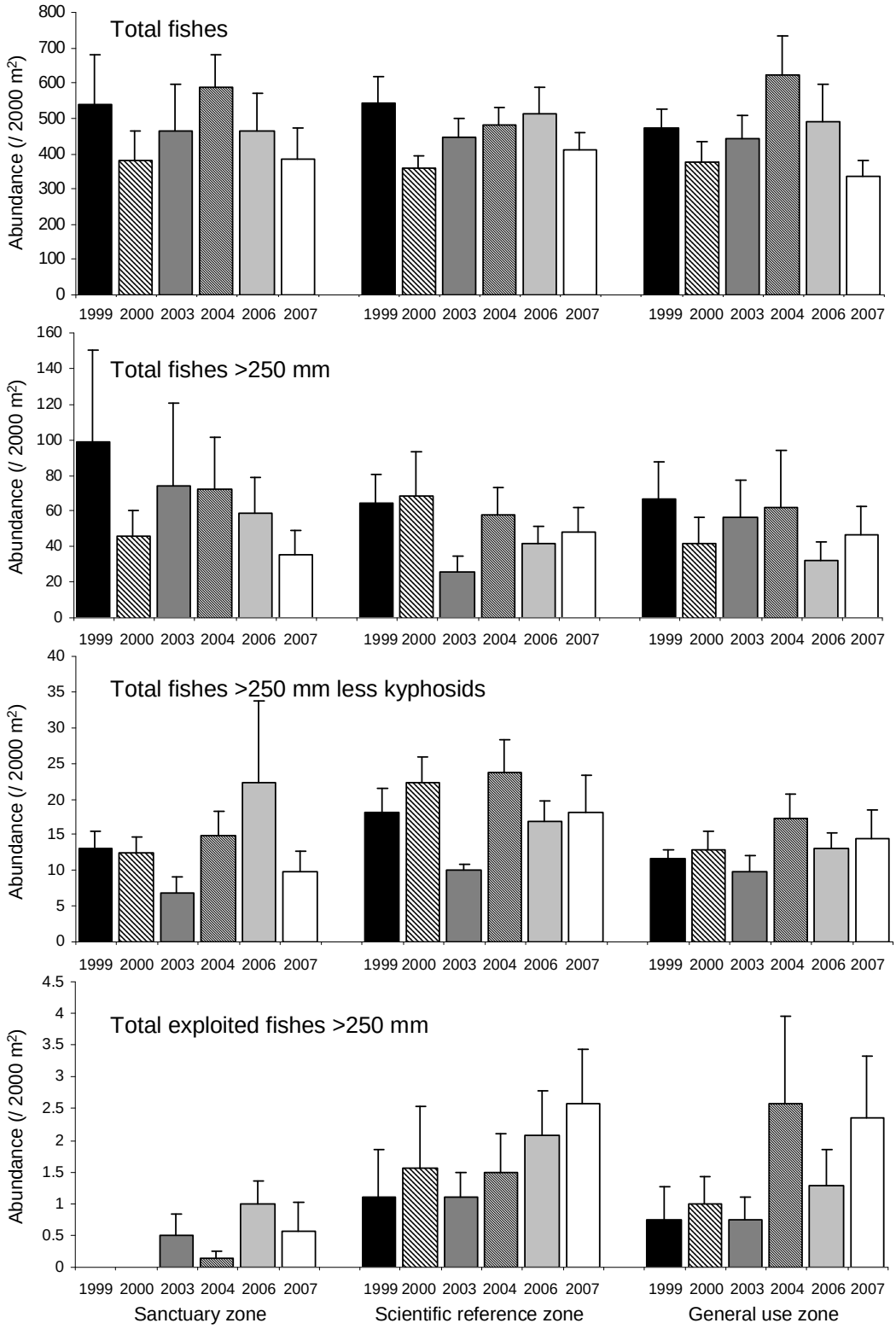

Figure 13. Mean total numbers (± SE) of fishes observed per site in different management zones in different survey years.

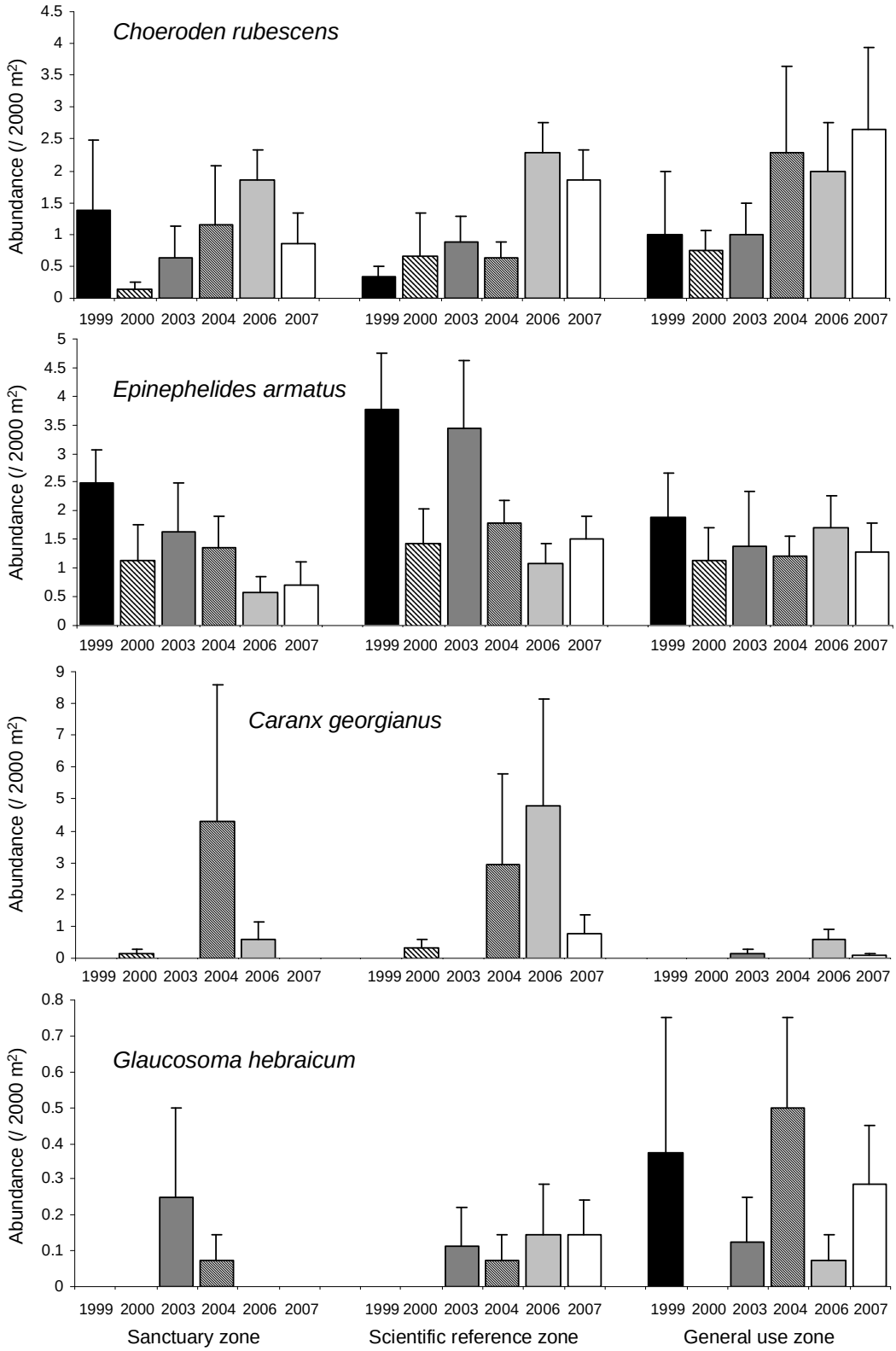

Figure 14. Mean total numbers (± SE) of exploited fishes observed per site in different management zones in different survey years.

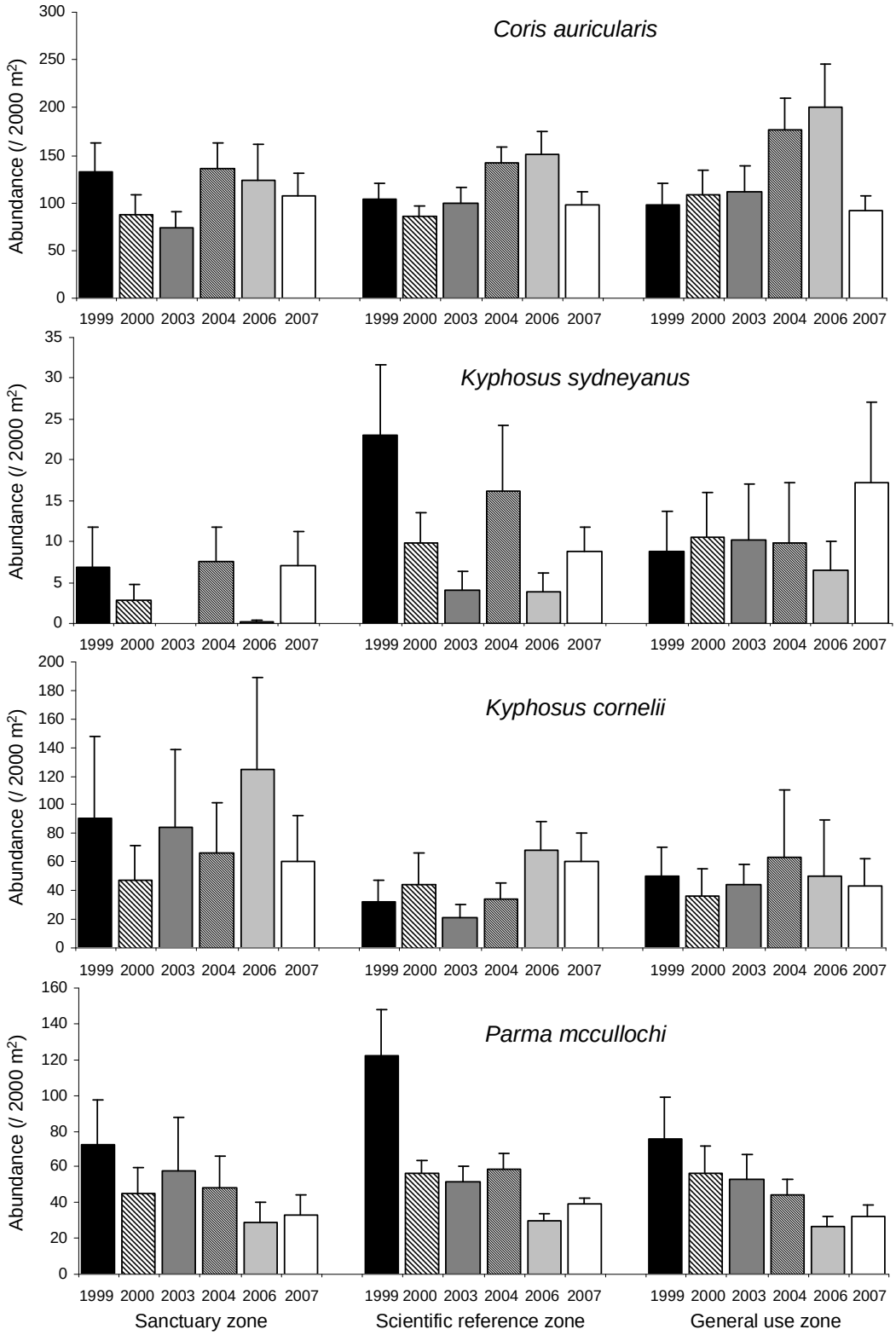

Figure 15. Mean total numbers (± SE) of common unexploited fishes observed per site in different management zones in different survey years.

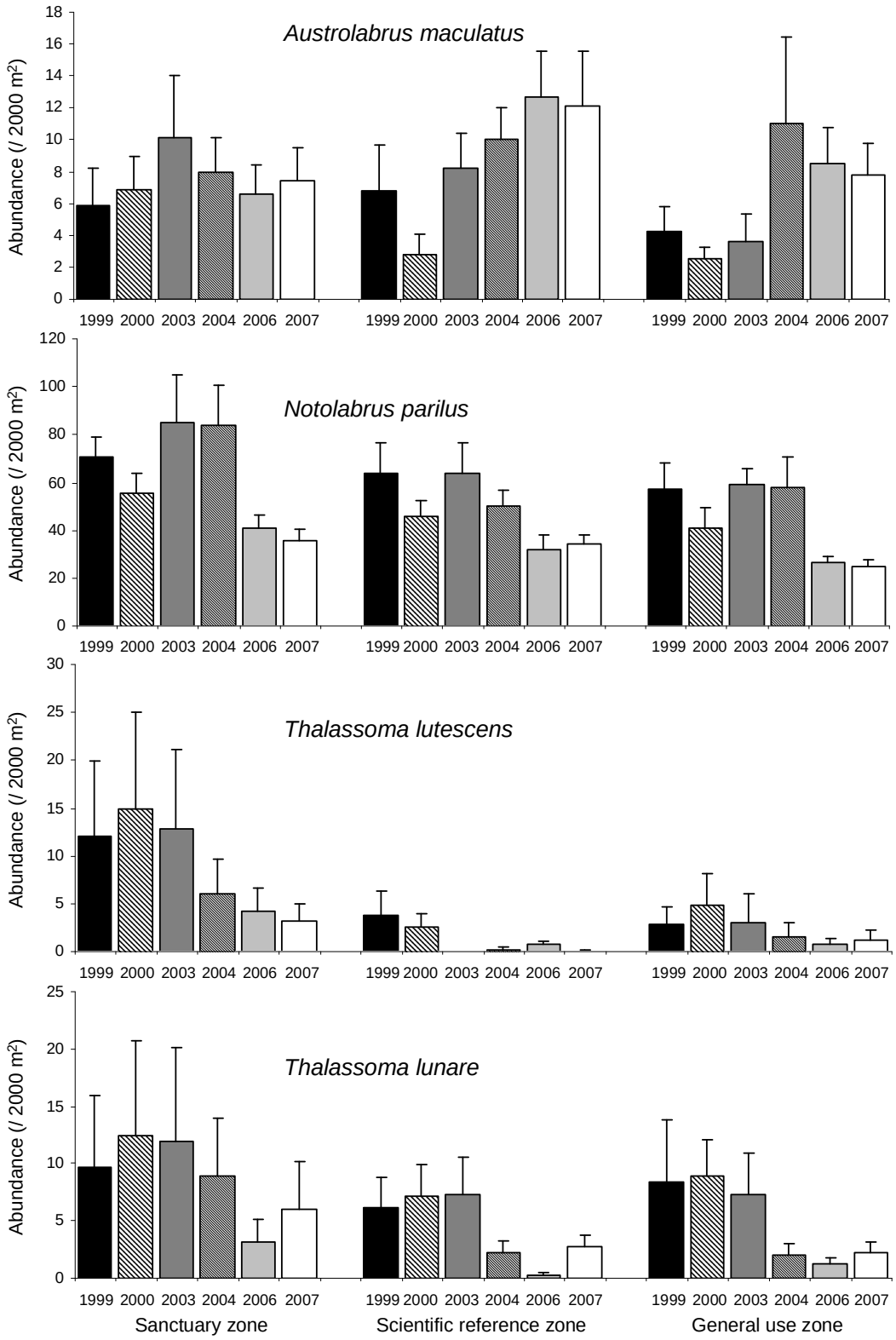

Figure 16. Mean total numbers (± SE) of common unexploited fishes observed per site in different management zones in different survey years.

Table 7. Mean squares (MS), F-value (F) and significance (*: 0.05>p>0.01; **: 0.01>p>0.001; ***: 0.001>p) resulting from ANOVA using time and MPA effect as factors, and zone location as blocking factor for abundances of all fishes, large (>250 mm) fishes, exploited fishes, and common fish species. Data were log transformed). Degrees of freedom are 1 (time), 1 (MPA effect), 6 (zone location) and 74 (error).

Taxon	Time		MPA		Zone location		Time x MPA		Error
	MS	F	MS	F	MS	F	MS	F	MS
Number of fishes	0.85	3.11	1.08	3.96*	0.35	1.28	0.40	1.48	0.27
Number of large fishes	1.22	0.98	4.84	3.88	2.49	1.99	0.27	0.21	1.25
Large fishes less kyphosids	1.39	2.45	0.41	0.72	1.32	2.34 *	0.21	0.38	0.57
Large exploited fishes	0.15	0.31	0.20	0.40	1.52	3.14 **	0.00	0.00	0.48
Choerodon rubescens	2.56	5.95*	0.90	2.10	0.95	2.21	0.02	0.05	0.43
Epinephelides armatus	1.23	3.77	0.15	0.46	0.47	1.43	0.02	0.07	0.33
Caranx georgianus	0.79	1.57	0.30	0.59	0.66	1.33	1.08	2.16	0.50
Glaucosoma hebraicum	0.04	0.57	0.15	2.13	0.07	0.91	0.02	0.25	0.07
Austrolabrus maculatus	0.00	0.00	2.04	2.25	2.28	2.51 *	1.00	1.10	0.91
Coris auricularis	0.25	0.54	3.25	7.07**	0.73	1.58	0.03	0.06	0.46
Notolabrus parilus	7.28	37.10***	0.01	0.03	0.67	3.42 **	0.15	0.77	0.20
Thalassoma lunare	0.28	0.32	5.34	6.11*	1.18	1.35	0.11	0.13	0.87
Thalassoma lutescens	3.88	3.66	3.55	3.35	2.32	2.18	0.02	0.02	1.06
Kyphosus cornelii	3.97	1.21	5.41	1.65	8.76	2.67 *	7.04	2.14	3.29
Kyphosus sydneyanus	0.34	0.19	0.67	0.38	2.02	1.14	2.30	1.30	1.77
Parma mccullochi	3.70	3.86	2.94	3.07	2.13	2.22 *	0.00	0.00	0.96

Table 8. Spearman rank correlations relating change in abundances of all fishes, large (>250 mm) fishes, exploited fishes, and common fish species following protection from fishing with distance of site from protected zone (= sanctuary zone + scientific reference zone) boundary and with size of protected zone; *: 0.05>p>0.01.

Taxon	Distance to protected area boundary	Area protected
Number of fishes	0.16	0.24
Number of large fishes	-0.10	0.00
Number of large fishes less kyphosids	-0.15	0.05
Number of large exploited fishes	0.02	0.28 *
Choerodon rubescens	0.10	0.15
Epinephelides armatus	-0.21	-0.28
Caranx georgianus	0.08	-0.03
Glaucosoma hebraicum	0.20	0.34 *
Austrolabrus maculatus	0.16	0.20
Coris auricularis	-0.02	0.07
Notolabrus parilus	0.07	0.03
Thalassoma lunare	0.04	-0.13
Thalassoma lutescens	-0.04	-0.05
Kyphosus cornelii	-0.02	0.06
Kyphosus sydneyanus	-0.13	-0.12
Parma mccullochi	-0.31 *	-0.04

Fish biomass

Estimated total fish biomass at sites was stable through time and between different zone locations (Fig. 17), with no significant effects indicated by ANOVA (Table 9). The estimated biomass of several trophic groups of fishes did, however, change at the regional level through time (Table 9). The estimated biomass of planktivores and higher carnivores both increased through time, whereas the biomass of benthic carnivores declined (Fig. 18). Notably, when pelagic species were removed from calculations of higher carnivore biomass, no change after the enforcement fishing prohibition, was indicated. Thus, large pelagic fish species contributed most of the temporal change in biomass of the higher carnivores.

No MPA related change over time was evident in any of the community metrics related to total fish biomass (Tables 9 and 10). The biomass of benthic carnivores exhibited considerable spatial variation, both between management zone types and between zone locations (Table 9). The biomass of higher carnivores also varied with zone location.

Restrictions on fishing should increase survival rates of large individuals in exploited stocks, shifting the size-distribution and increasing mean size of fishes in a population. This was assessed by calculating correlations between change in mean biomass of individual fishes in protected areas following restrictions on fishing and both distance from protected area boundary and protected area size. ANOVA was not used because of the large number of zero values and variance structure that was highly heterogeneous.

Breaksea cod (*Epinephelides armatus*) showed MPA related effects for both these sets of correlations, with mean observed size increasing significantly with distance from protected area boundary and also with size of protected area (Table 11). The mean biomass per individual increased by 20% in protected zones following restrictions on fishing, and decreased 20% in general use zones (Fig. 19). Silver trevally (*Caranx georgianus*) and moon wrasse (*Thalassoma lunare*) also showed significant correlations between change in mean fish size and size of protected location, while buffalo bream (*Kyphosus sydneyanus*) showed a significant decline in mean size in protected locations (Table 11).

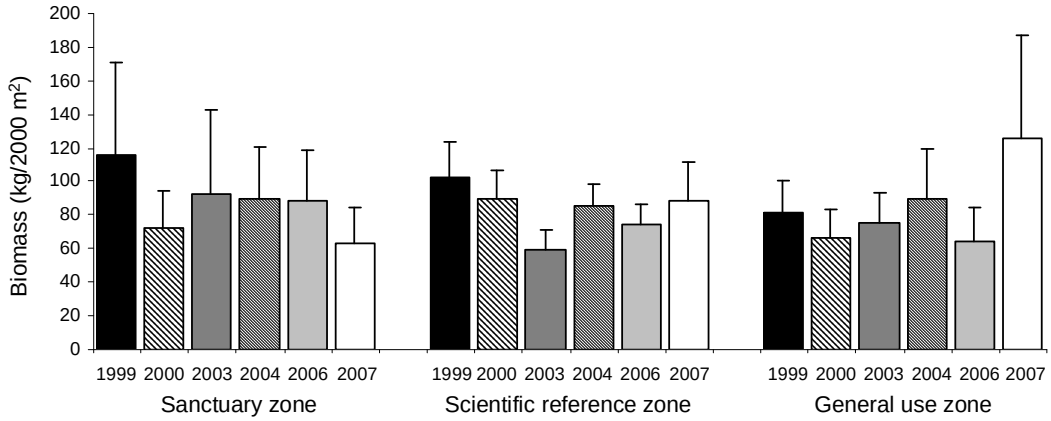

Figure 17. Mean total biomass (± SE) of fishes in different management zones in different survey years.

Table 9. Mean squares (MS), F-value (F) and significance (*: 0.05>p>0.01; **: 0.01>p>0.001; ***: 0.001>p) resulting from ANOVA using time and MPA effect as factors, and zone location as blocking factor for fish biomass. Data were log transformed). Degrees of freedom are 1 (time), 1 (MPA effect), 6 (zone location) and 74 (error).

Taxon	Time		MPA		Zone location		Time x MPA		Error
	MS	F	MS	F	MS	F	MS	F	MS
Herbivores	0.304	0.218	3.216	2.300	2.850	2.039	1.772	1.267	1.398
Planktivores	2.143	4.569 *	0.371	0.791	0.588	1.255	0.098	0.208	0.469
Benthic carnivores	2.850	11.538 ***	1.878	7.605 **	0.750	3.036 **	0.022	0.090	0.247
Higher carnivores	8.961	8.539 **	0.299	0.285	2.847	2.713 *	0.043	0.041	1.049
Higher canivores less pelagics	2.389	3.437	0.014	0.021	1.222	1.759	0.288	0.415	0.695
Total fishes	0.280	0.378	2.389	3.219	1.270	1.711	0.331	0.446	0.742

Table 10. Spearman rank correlations relating change in fish biomass per site for major trophic groups following protection from fishing with distance of site from protected zone (= sanctuary zone + scientific reference zone) boundary and with size of protected zone. None of the correlations were significant at p = 0.05.

Taxon	Distance to protected area boundary	Area protected
Herbivores	-0.18	-0.20
Planktivores	-0.07	0.11
Benthic carnivores	0.01	-0.08
Higher carnivores	-0.08	-0.02
Higher canivores less pelagics	-0.06	0.14
Total fishes	-0.18	-0.15

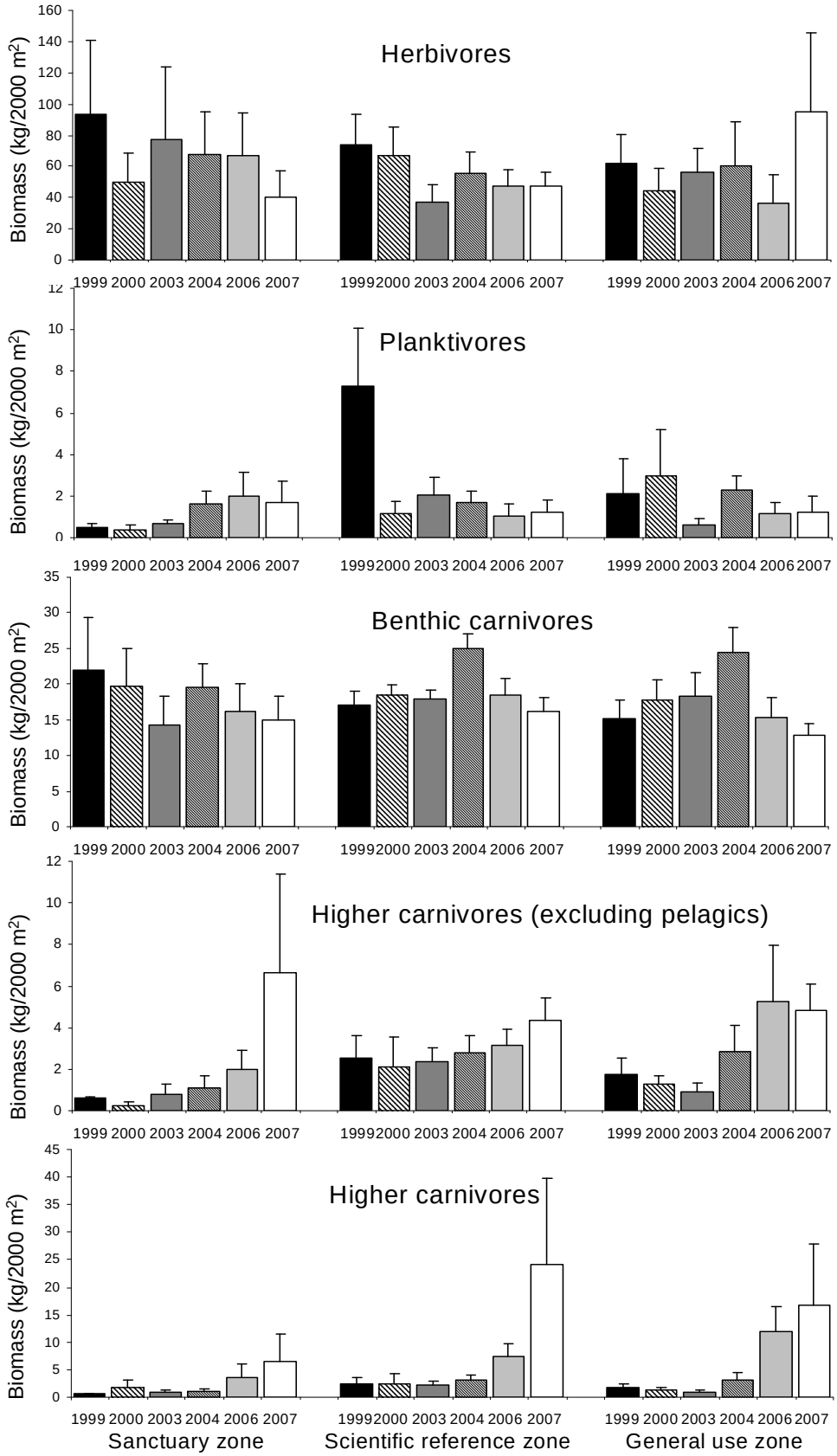

Figure 18. Mean biomass (± SE) of major trophic groups of fishes in different management zones in different survey years.

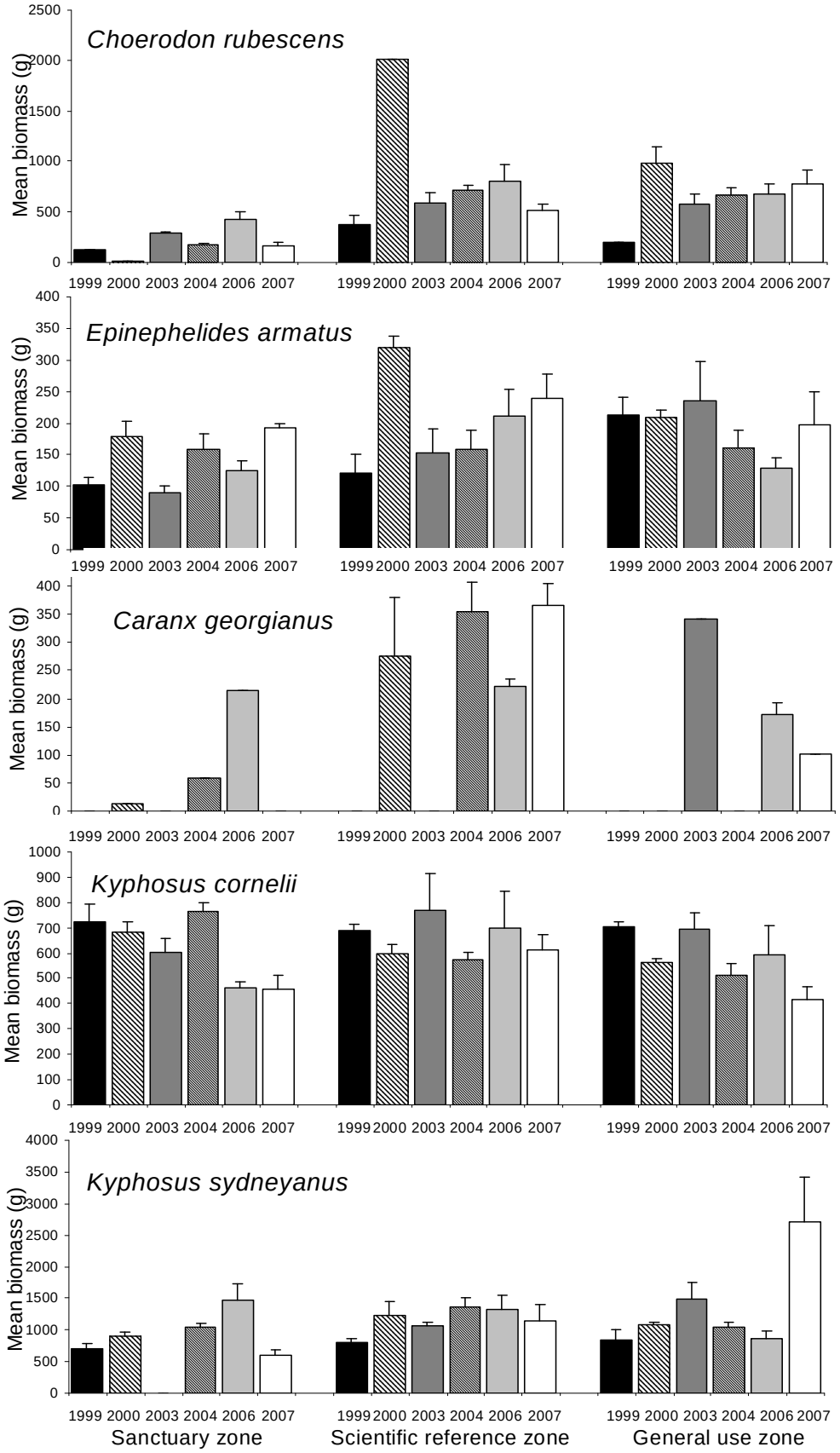

Figure 19. Mean estimated biomass per fish (± SE) for exploited and large non-exploited fishes in different management zones in different survey years.

Table 11. Spearman rank correlations relating change in mean size of fish of common species following protection from fishing with distance of site from protected zone boundary and with size of protected zone (*: 0.05>p>0.01; **: 0.01>p>0.001).

Taxon	Distance to protected area boundary		Area protected	
Caranx georgianus	0.16		0.43	**
Choerodon rubescens	0.07		0.04	
Epinephelides armatus	0.38	**	0.40	**
Glaucosoma hebraicum	-0.28		-0.32	
Austrolabrus maculatus	-0.08		-0.13	
Coris auricularis	-0.10		0.00	
Notolabrus parilus	-0.05		0.09	
Thalassoma lunare	0.18		0.31	*
Thalassoma lutescens	-0.27		0.08	
Kyphosus cornelii	0.10		0.19	
Kyphosus sydneyanus	-0.33	*	-0.38	*
Parma mccullochi	-0.07		0.01	

Mobile macro-invertebrate abundance

Compared to fish abundance, the abundance of macro-invertebrates varied greatly across the JBMP, with most of the common species exhibiting highly significant variation between zone locations. Nevertheless, macro-invertebrate densities were stable through time and no regional change over time was evident for any of the common species examined (Fig. 21; Table 12). No MPA-related changes were detected in ANOVAs (Table 12) and rank correlations (Table 13), including the analyses involving rock lobsters.

Thus, although densities of rock lobsters were predicted to increase in sanctuary zones relative to fished zones (scientific reference and general use), no such increase was statistically apparent (Fig. 21). Rock lobster numbers did, however, show a slight overall increase in sanctuary zones following protection (from 8.2 to 8.6 per 200 m^2), whereas numbers in fished zones (scientific reference and general use) showed a marked decline (from 4.4 to 2.6 per 200 m^2). The lack of statistical significance relates to the highly patchy distribution of rock lobsters, particularly amongst sanctuary zones, which include one anomalous site (Inner Boullanger Island) producing about 30% of the total numbers of rock lobsters recorded across all sites.

Mean size of rock lobsters provided a more stable signal through time (Fig. 22). Contrary to predictions, mean rock lobster carapace length increased in fished zones relative to unfished zones following protection, from a mean of 49 mm to 69 mm in scientific reference and general use zones between the periods 1999-2004 and 2006-07, compared to a change from 59 mm to 63 mm in sanctuary zones over the same period (Figs. 22 and 23). Change in mean size between these two periods was significantly negatively correlated with distance from sanctuary zone boundary (r_s = -0.421; 0.01>p>0.001), but not with size of sanctuary zone (r_s = -0.294; 0.05>p).

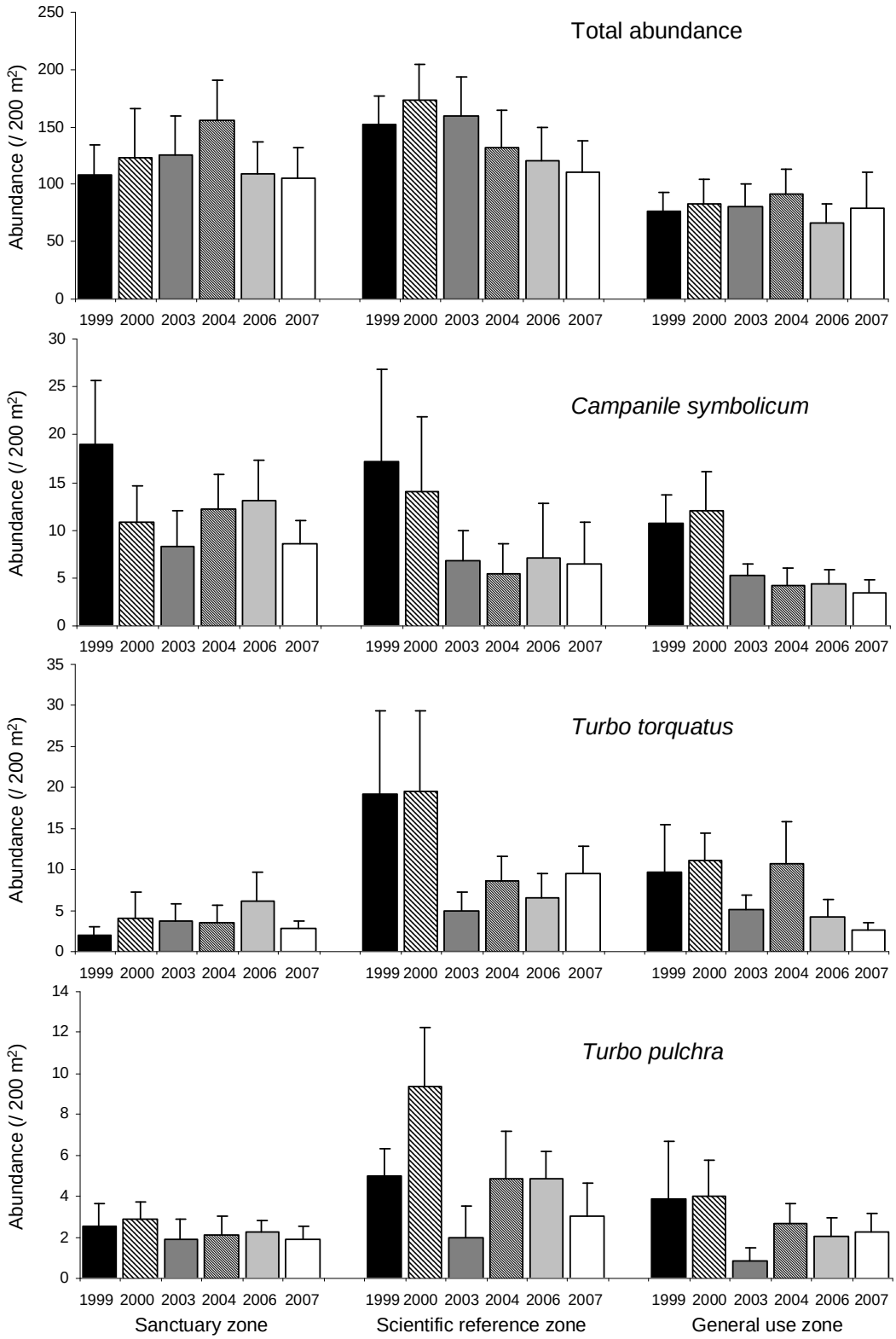

Figure 20. Mean total numbers (± SE) of mobile macro-invertebrates and common mollusc species observed per site in different management zones in different survey years.

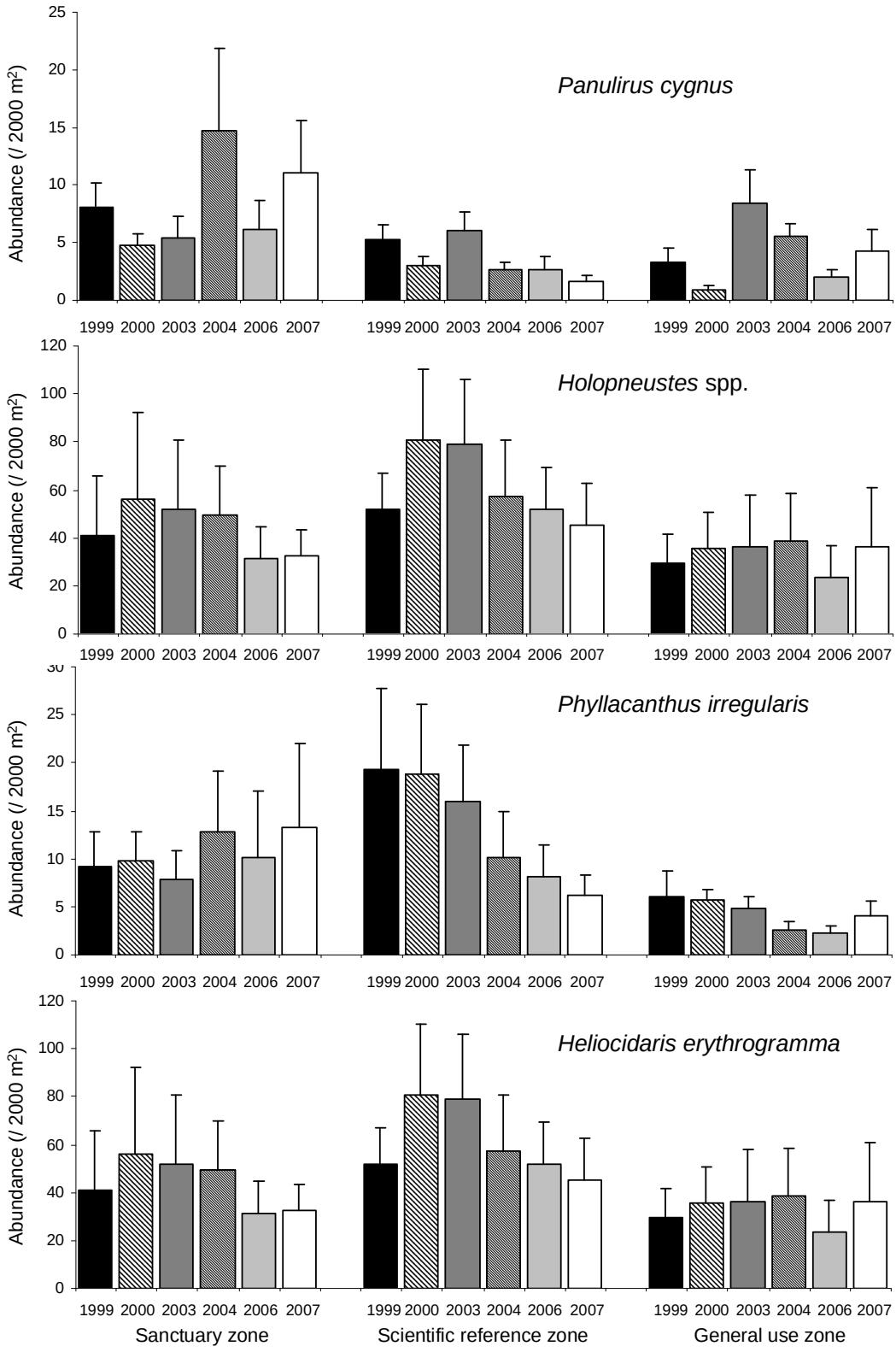

Figure 21. Mean total numbers (± SE) of rock lobsters and common echinoid species observed per site in different management zones in different survey years.

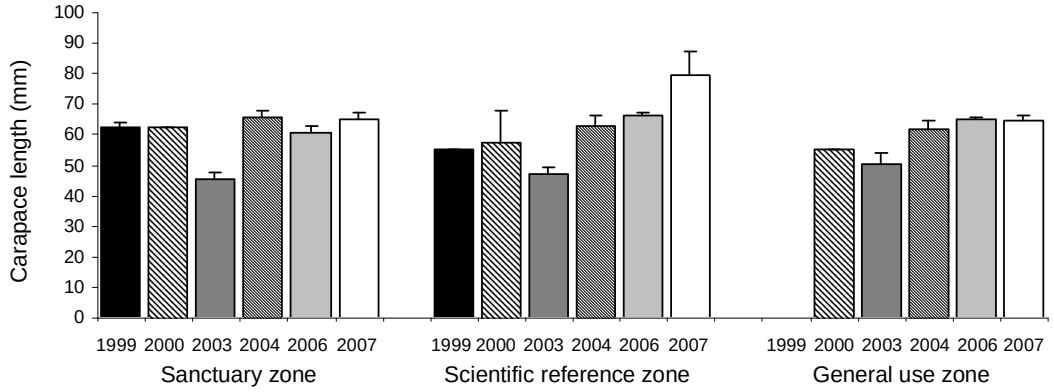

Figure 22. Mean carapace length (± SE) of rock lobsters in different management zones in different survey years.

Table 12. Mean squares (MS), F-value (F) and significance (*: 0.05>p>0.01; **: 0.01>p>0.001; ***: 0.001>p) resulting from ANOVA using time and MPA effect as factors, and zone location as blocking factor for mobile macro-invertebrate abundance. Data were log transformed). Degrees of freedom are 1 (time), 1 (MPA effect), 6 (zone location) and 74 (error).

Taxon	Time		MPA			Zone location			Time x MPA		Error
	MS	F	MS	F		MS	F		MS	F	MS
Total macro-invertebrates	0.83	1.27	0.28	0.42		2.34	3.55	**	0.04	0.06	0.66
Campanile symbolicum	0.14	0.10	0.05	0.03		2.80	1.88		0.00	0.00	1.49
Turbo pulchra	0.10	0.18	0.07	0.12		1.39	2.39	*	0.12	0.21	0.58
Turbo torquatus	0.00	0.00	0.46	0.45		4.71	4.55	***	0.68	0.66	1.04
Panulirus cygnus	1.64	1.46	0.53	0.48		1.23	1.10		0.16	0.14	1.12
Holopneustes spp.	1.47	2.05	4.92	6.84	*	2.31	3.20	**	0.80	1.11	0.72
Phyllacanthus irregularis	1.69	1.84	1.00	1.09		5.77	6.30	***	0.42	0.45	0.92
Heliocidaris erythrogramma	0.11	0.05	0.02	0.01		13.33	6.61	***	0.47	0.23	2.02

Table 13. Spearman rank correlations relating change in mobile macro-invertebrate abundance per site following protection from fishing with distance of site from protected zone (= sanctuary zone + scientific reference zone) boundary and with size of protected zone. None of the correlations were significant at p = 0.05.

Taxon	Distance to protected area boundary	Area protected
Total macro-invertebrates	0.04	-0.03
Campanile symbolicum	0.06	0.04
Turbo pulchra	-0.15	-0.02
Turbo torquatus	-0.01	0.09
Panulirus cygnus	-0.01	0.01
Holopneustes spp.	0.03	0.09
Phyllacanthus irregularis	-0.18	-0.21
Heliocidaris erythrogramma	0.17	0.06

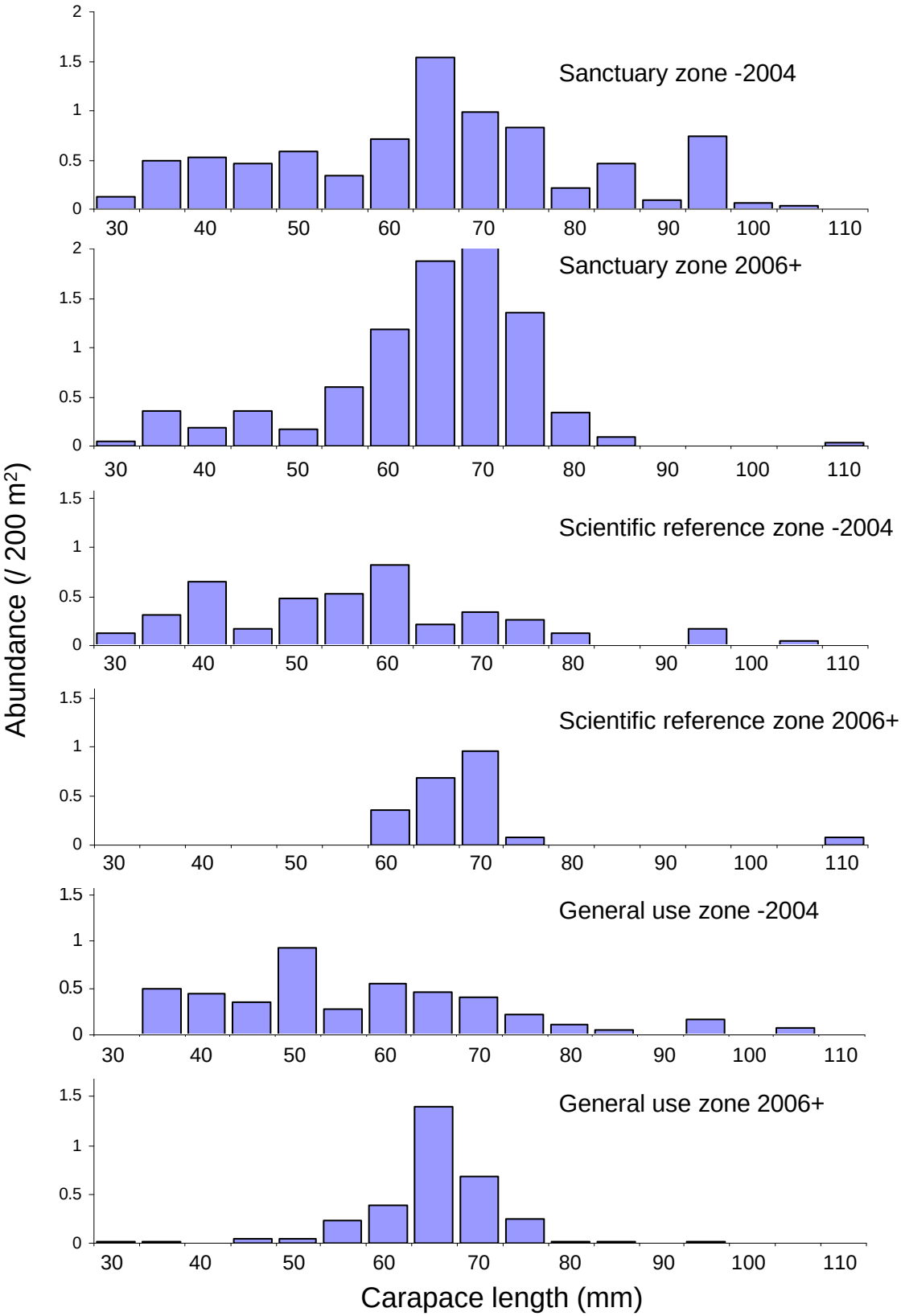

Figure 23. Mean numbers of rock lobsters in different size-classes in different management zones before (-2004) and after (2006+) the introduction of restrictions on fishing.

Macroalgal and sessile invertebrate cover

Coral cover differed significantly between zone types (Table 14), with five times higher mean cover in sanctuary zones (1.8%) than in scientific reference zones (0.37%) and general use zones (0.39%) (Fig. 24). Coral cover also differed between zone locations (Table 14), with highest cover near Fishermans Island.

Amongst the macroalgae, cover of *Sargassum* spp. and foliose red macroalgae varied significantly between zone types (Table 14), with highest cover of foliose red algae and lowest cover of *Sargassum* spp. in sanctuary zones (Figs. 24 and 25). Filamentous red algae decreased through time, several taxa (*Ecklonia radiata*, *Caulerpa* spp. and filamentous red algae) varied with zone location, but no taxon showed significant MPA-related changes with time (Tables 14 and 15).

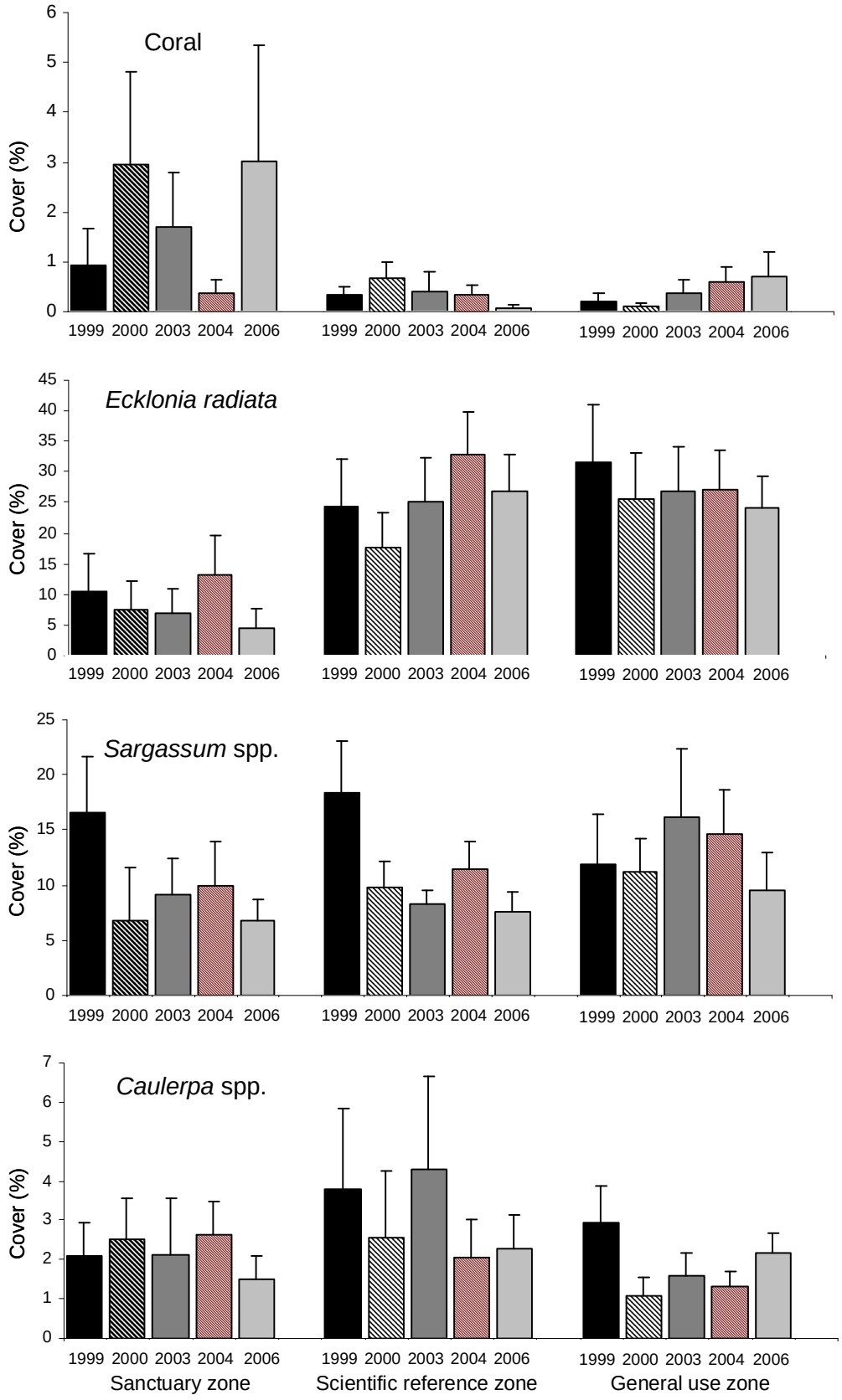

Figure 24. Mean total cover (± SE) of scleractinian corals and common foliose plant taxa per site in different management zones in different survey years.

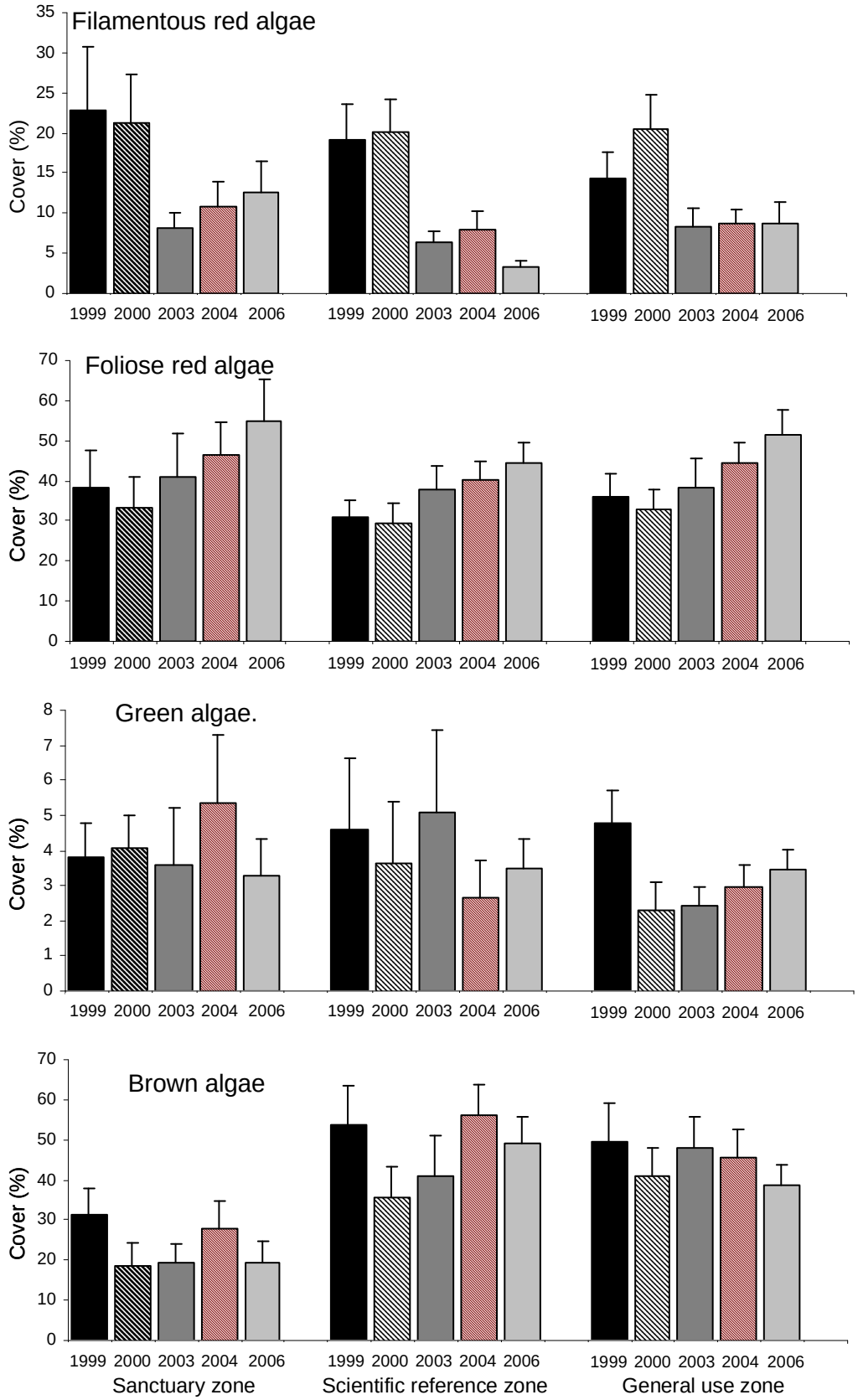

Figure 25. Mean total cover (± SE) of major foliose plant taxa in different management zones in different survey years.

Table 14. Mean squares (MS), F-value (F) and significance (*: 0.05>p>0.01; **: 0.01>p>0.001; ***: 0.001>p) resulting from ANOVA using time and MPA effect as factors, and zone location as blocking factor for scleractinian corals and macroalgal taxa. Data were log transformed). Degrees of freedom are 1 (time), 1 (MPA effect), 6 (zone location) and 69 (error).

Taxon	Time			MPA			Zone location			Time x MPA		Error
	MS	F		MS	F		MS	F		MS	F	MS
Coral	0.02	0.05		3.28	10.52	**	0.87	2.78	*	0.02	0.05	0.31
Ecklonia radiata	0.38	0.17		5.47	2.47		6.77	3.06	**	0.01	0.00	2.21
Sargassum spp.	0.63	0.83		7.54	9.93	**	1.28	1.69		2.10	2.76	0.76
Caulerpa spp.	0.01	0.03		1.01	2.66		1.27	3.35	**	0.43	1.13	0.38
Filamentous red algae	7.21	8.49	**	0.36	0.42		2.26	2.67	*	0.38	0.44	0.85
Foliose red macroalgae	0.59	1.84		4.39	13.80	***	0.42	1.33		0.00	0.01	0.32
Green macroalgae	0.04	0.09		0.04	0.07		0.71	1.53		0.02	0.04	0.47
Brown macroalgae	0.00	0.00		0.08	0.09		1.82	2.06		0.36	0.40	0.89

Table 15. Spearman rank correlations relating change in percent cover of scleractinian corals and macroalgal taxa following protection from fishing with distance of site from protected zone (= sanctuary zone + scientific reference zone) boundary and with size of protected zone. None of the correlations were significant at p = 0.05.

Taxon	Distance to protected area boundary	Area protected
Coral	-0.08	0.08
Ecklonia radiata	0.21	-0.03
Sargassum spp.	-0.01	-0.10
Caulerpa spp.	-0.07	-0.14
Filamentous red algae	-0.22	-0.21
Foliose red macroalgae	0.06	0.03
Green macroalgae	0.06	-0.07
Brown macroalgae	0.16	0.05

4. Discussion and recommendations

4.1 Reef monitoring rationale

The creation of a mosaic of management zones in the seascape through the declaration of MPAs represents an ecological experiment involving the exclusion of human predators at a vast spatial scale (Walters and Holling 1990). The JBMP monitoring method was developed to capitalise on this experiment (Edgar and Barrett 1999). It involves underwater visual census of densities of fishes, invertebrates and plants along 200 m transects at replicate sites to quantify biological changes in response to the introduction and enforcement of fishing restrictions in different management zones.

We consider that visual census techniques provide the most effective technique for monitoring species at shallow-water sites in MPAs because they are non-destructive and permit the collection of large amounts of data on a broad range of species within a short dive period. MPA monitoring programs need to cover a range of taxa because, in addition to heavily-exploited species that are predicted to recover in new MPAs, significant secondary effects of fishing are known to also occur that would otherwise go undetected (Babcock *et al.* 1999).

The overriding consideration when planning the monitoring design was that temporal change in the different management zone treatments provided the primary focus of study. Consequently, spatial variation at the site level that interferes within the detection of the temporal signal was minimised as much as possible. This was done by censusing fixed sites through time, surveying species along set depth contours, sampling in the same season in different years, and aggregating data over a long distance (200 m) per site to smooth fine scale variation.

To minimise natural variation and seasonal effects, sites are fixed and sampled each year over similar dates from mid October to early November. The 200 m transect distance is subdivided into four contiguous 50 m long blocks, each of which is 10 m wide for censuses of mobile fishes and 1 m wide for censuses of mobile macro-invertebrates and cryptic fishes. In addition, macrophytes and sessile invertebrates are surveyed 20 times with a 0.25 m^2 quadrat at 10 m intervals along the transect line.

This fixed 'extended-transect' sampling design was selected to maximise the amount of information gathered at each site by three divers, each with a single tank of air. Three sites can be surveyed per day, weather conditions permitting. Pilot trials indicated that if divers reduced the amount of information collected per site, for example by surveying two rather than four 50 m long blocks, then site coverage would not have increased greatly because of the lengthy time required to move between sites (pull anchor, gear up for diving, set transect lines etc). Collection of additional information at each site would require either a second dive team or reduced site coverage.

The collection of data from four 50 m long blocks is best viewed as an approach to increase the precision of estimates of mean values for a 50 m block at a site. Information on spatial substructure within sites, in the form of data from the four contiguous 50 m-long transects, was not used to assess variance within sites. Rather the 200 m transect was subdivided into four blocks because:

1. Data are more easily compared with results of other investigators, who often use transect lengths of 50 m.

2. Different divers can collect information in different 50 m sections of the 200 m length, allowing equitable distribution of dive time regardless of number of divers, and permitting analysis of observer (between diver) effects.

3. If greater precision at a site is required, for example if rock lobster numbers are highly spatially-variable but are a management priority, then extra 50-m blocks can be added. Similarly, the number of 50-m blocks can be reduced if dive time is limited, such as when surveying deep sites. In both cases, data at the 50-m block scale remain directly comparable with data for other sites.

The extended-transect design represents a compromise between power and generality, lying intermediate along the spectrum from more general site studies that involve random replicate transects at each site, and more powerful studies with a single fixed-transect permanently anchored to the seabed.

The extended-transect design is considerably more powerful than a random-transect design, but with less generality in associated statistical tests. Although an understanding of within-site variation can be critical for studies with other aims, individual sites had no intrinsic

importance in this MPA study. Our interest was focused on within- and between-zone effects, with sites providing replicate information for analyses. Zone locations were regarded as important in their own right because of a management need to identify whether particular zones were ineffective with respect to management aims, such as if the zone was too small or illegal exploitation affected population numbers. However, because the general level of ecological response to the new MPA zones that could be identified after two years of protection was low, data pertaining to individual zones have not been presented.

Advantages of random-transect methods over the extended-transect method are: (i) sites encompass a greater total area of seabed because a range of depths are surveyed at each site rather than a single depth contour, thus increasing generality, and (ii) information is gathered on spatial variance within sites. However, for a study of MPA effects, we considered that these advantages were greatly outweighed by disadvantages, which include: (i) spatial noise associated with randomised placement of transects that obscures the fundamental temporal signal, (ii) lost diving time during periods when divers move to the start of different replicate transects, resulting in reduced data collection per unit time, (iii) difficulties in truly randomising transect placement and spatial biases associated with haphazard placement, and (iv) confounding with depth as a consequence of some sites being relatively flat with little depth range, and others being steeply-sloping and encompassing a large depth range. We regard depth was better to be included as an explicit variable within analyses rather than contributing to spatial noise between replicates.

A design involving transects that are permanently attached to the seabed would be more powerful at detecting temporal effects than our design, but at some minor cost in generality and at considerable extra cost in dive time. The cost in generality for a physically-fixed transect design relates to the fact that our transects were relocated on each sampling event within a band that extended ca 1 m in depth (due in large part to different tidal heights at the time of each survey) and ca 20 m in horizontal extent (due to imprecision in site relocation). Thus, some spatial ‘noise’ is added to the temporal ‘signal’ in our design, reducing power but also reducing the possibility that overall conclusions are affected by anomalous positioning of a transect.

Disadvantages of utilising a physically-fixed transect are threefold. Firstly, the presence of a permanent transect line could affect survey results. For instance wave-induced movements of the line may abrade plants and potentially affect the habitat and thus the ecosystem components censused along the transect. Secondly, management and social issues arise with installation of fixed infrastructure. This includes a reduction in aesthetic values associated with diving in MPAs, given that 200 m long ropes or chains permanently attached to the seabed in sanctuary zones, or permanent markers, would represent a visual intrusion to recreational divers. Thirdly, despite the theoretical increase in power to detect temporal signal for physically-fixed transect designs, power is adversely affected in a practical sense by reduced replication. Considerable dive time and cost is required to initially set up permanent transect lines and seabed markers. If transect lines are left attached between surveys, then they need maintenance, perhaps with replacement after two or three years. If lines are strung on each survey between permanent markers such as star picket posts, then dive time is reduced by the extra time required to set the line after locating markers, some of which may disappear between annual surveys.

4.2 Natural spatial and temporal variation in the JBMP

The JBMP provides home for a diverse range of fishes, macro-invertebrates, macroalgae and seagrasses, a large proportion of which are endemic to southwestern WA. A few corals near the southern end of their ranges were recorded on transects in JBMP; however, reef habitats are much more characterised by warm-temperate seaweeds rather than by corals. Subtropical fishes and corals in the JBMP were most evident in the Fishermans Island Sanctuary Zone, a shallow protected reef system in the northern outer lagoonal region where water is clear and temperatures rise on sunny days.

Compared to other locations investigated as part of the broader temperate Australian MPA study, a notable characteristic of the JBMP is a paucity of planktivorous fishes. The mean observed biomass of planktivores was ~2 kg per site, which compares with a mean of ~50 kg per 2000 m^{-2} in the Lord Howe Island Marine Park (Edgar *et al.* 2008a), the only region where comparative analyses have been undertaken to date. The low biomass of planktivores presumably relates ultimately to extremely low nutrient conditions that prevail across the region (Bancroft 2005; Babcock *et al.* 2006), and an associated paucity of plankton prey (Department of Conservation and Land Management 2005). The mean biomasses of herbivorous, benthic carnivorous and higher carnivorous fishes do not differ greatly in general use zones between the Jurien Bay and Lord Howe Island Marine Parks.

The major factor that affects the distribution of biotic communities in the JBMP is the offshore gradient. Plants and animals associated with the sheltered inshore reefs differ greatly from those on outer reefs. Within the set of sheltered reef habitats, substantial biological differences are also evident between reef communities in the Cavanagh and Grey Sanctuary Zone regions in the south, and reefs in the Fishermans Island Sanctuary Zone in the north. These differences in community structure are comparable in magnitude to differences between sheltered and offshore reefs (see PCA results in Fig. 2). By contrast, offshore reefs were found to be highly homogeneous, with little variation in community structure apparent across the full extent of the JBMP.

The floral and faunal communities at different locations generally exhibited a low level of change between years compared to variation between sites. MDS trajectories depicting change between early and recent surveys showed little overlap between different zone locations (Fig. 3). Nevertheless, the magnitude of observed temporal change varied between taxonomic groups, with most change evident amongst the sessile biota (Fig. 8), and least change evident amongst the fishes (Fig. 5).

The lack of survey data for sessile biota in 2007 presumably contributed to the relatively high magnitude of temporal change at different locations for this taxonomic group. Averaging between the years 2006 and 2007, as was done for the fish and macro-invertebrate data sets, would reduce spatial noise in data and provide a more stable temporal signal.

Some of the interannual variation in species richness of the sessile biota also probably resulted from variation in the taxonomic skills of divers when assessing cover of macroalgae. In contrast to fishes and mobile invertebrates, where all animals are recorded to the species level, seaweed species are grouped within higher taxa whenever a macroalga cannot be identified to species. This occurs when the species is unknown to the diver or when reproductive structures critical for identification are absent. Through time, as knowledge of

the algal flora increases, additional species are recognised *in situ* by divers and recorded at the species level, rather than lumped into a higher taxon.

Although we tried to minimise the effect of diver knowledge by using only three experienced divers for assessing macroalgae during the five survey years, the influence of experience possibly contributed to trends in macroalgal species richness, with increasing total numbers of macroalgae recorded through time (Fig. 12). Increasing macroalgal richness was due to increasing numbers of red algal species observed (Fig. 12; Table 5), an expected outcome of increased diver knowledge given that red macroalgae comprise the floral group where taxonomic skill is most important and where additional species will be recognised with experience.

Regardless, relatively few of the trends through time can be attributed to biases in diver training. In addition to foliose red algal species richness, red algal cover also increased through time (Fig. 24), and this measure is not influenced by diver experience. Moreover, zone locations with relatively large temporal changes in fish abundance and mobile macro-invertebrate abundance trended in a similar direction in MDS plots, indicating a coherent region-wide change in ecological communities. Consistent change across the region is also evident in plots showing abundance trends of individual fish species, including long-term population decline for the wrasses *Notolabrus parilus*, *Thalassoma lutescens* and *Thalassoma lunare* (Fig. 16).

4.3 Effectiveness of JBMP zoning scheme and monitoring program

Identification of effects of fishing over the past decade in the JBMP is complicated by the long-term regional ecological change that occurred over the period of monitoring, and also by the non-random distribution of zone types. As clearly evident in MDS plots, sanctuary zones were often outliers with respect to floral and faunal communities, and consequently lacked good control locations to account for regional interannual trends in data that were unrelated to effects of fishing. In contrast to sanctuary zone locations, reef communities in the three scientific reference zone locations were very similar to each other, and also to communities at offshore reference locations in the general use zone.

Very few observable ecological changes associated with new fishing restrictions in sanctuary and scientific reference zones have been identified to date in the JBMP monitoring program. None of the 49 ecological metrics examined using ANOVA was found to change significantly in protected zones relative to data obtained prior to fishing restrictions and at reference sites located in fished areas. Of 61 rank correlations assessed between ecological change and distance from protected area boundary, only three significant outcomes were observed, while six significant outcomes were found for the 61 rank correlations with size of protected area. Given use of an α-value of 0.05, which implies about three significant correlations that are spurious (Type 1 errors) amongst each set of 61 tests, then most of the significant results are probably a consequence of such errors.

Nevertheless, not all significant results are likely to be spurious given that four of the six significant correlations with protected area size involved exploited species, yet only 15 of the 61 metrics tested involved this subset of all species. Metrics significantly correlated with protected area size included mean size of breaksea cod (*Epinephelides armatus*) and silver trevally (*Caranx georgianus*), and the abundance of large (>250 mm) exploited fishes as a

group and also dhufish (*Glaucosoma hebraicum*). The trends in breaksea cod size and numbers of large exploited fishes are likely to be real, given adequate data that is consistent between years. By contrast, the trends in silver trevally size and dhufish abundance relate to highly patchy data, with the correlations best regarded as flags for confirmation or otherwise through the longer term.

The scarcity of observed ecological changes associated with the new JBMP zoning scheme may result from several possible causes: (i) low power in statistical design to detect change at $\alpha = 0.05$, (ii) low pre-existing level of fishing pressure in some of the more inaccessible locations, (iii) limited time since gazettal of fishing restrictions, (iv) lack of adequate enforcement of fishing restrictions, (v) MPA zones are ineffective in achieving biodiversity aims, or, most likely, (vi) a combination two or more of these factors.

The power of tests varies greatly between metrics, depending on patchiness of data and consistency of means through time. Data sets coherent between consecutive years (e.g. number of fish species recorded per year, abundance of *Notolabrus parilus*) should allow detection of relatively small effects, such as the 17% increase in mean log biomass per individual breaksea cod that was found significant at $p<0.01$. Note that year to year consistency is a better index of power than the standard error terms shown in figures, which relate to spatial variance between sites. Standard errors can be high but temporal tests of MPA effects will still have reasonable power provided that site to site to site variance is consistent between years.

Overall, a lack of statistical power does not appear to be a major issue with respect to the paucity of significant results. Plots of temporal changes in species richness and abundance indicate very little net change in mean values associated with different zones for the periods before and after restrictions on fishing. Thus, even if sampling effort had been greatly increased with much higher levels of site replication, few significant changes would have been noted over the two year period since enforcement of fishing restrictions.

A second explanation for the low level of observed change is that pre-existent fishing pressure across the region was low and historically depressed stocks only slightly, hence any recovery to the unfished state can only be minor and difficult to detect. This hypothesis is likely true for species not actively targeted or that do not occur as bycatch in the major fisheries.

Compared to heavily exploited regions, such as the vicinity of Perth, the remoteness of the Jurien region and low local human population densities have likely prevented major historical declines in fish stocks. Nevertheless, because populations of rock lobsters, dhufish and baldchin groper are heavily targeted across their full range, population numbers of these species, at least, were likely to have been substantially depressed within the JBMP. Dhufish, in particular, are a comparatively rare species that provide the primary target for recreational fishers, hence even low fishing pressure would substantially reduce population numbers.

Dhufish notably provided one of the few significant MPA-related responses observed in the study, with a significant correlation relating change at sites to area of protection from fishing. Dhufish numbers increased 13% in the two years following protection in sanctuary and scientific reference zones, compared to an overall 29% decrease over the same period in general use zones. A recent study indicates that commercial and recreational dhufish catch

rates for the Midwest has been reasonably static for the length of this monitoring program (Wise *et al.* 2007)

Rock lobsters, on the other hand, showed no increase in population numbers or mean size within sanctuary zones, and an increase in mean size in fished zones. This unexpected outcome probably relates, at least in part, to the biology of the species, and an offshore migration of animals at the stage in their life-cycle when they are recruiting to the fishery (Phillips 1983). If rock lobsters settle on reefs in inshore sanctuary zones at the post-puerulus stage, then leave for deeper reefs on reaching minimum legal size, animals in protected zones are not subjected to fishing pressure, and no MPA-related effect can be expected.

Nevertheless, it is also possible that, while the majority migrate, some individuals remain on or return to inshore reefs, and that insufficient time has elapsed for the presence of these animals to statistically affect survey data. Some animals above legal size were observed on sanctuary zone reefs, and studies 250 km south at Rottnest Island indicate that marine reserves on shallow reefs can provide effective sanctuaries for lobsters (Babcock *et al.* 2007).

The two years that has elapsed since restrictions on fishing were enacted is minor in relation to the life-span of local commercial species, hence major changes are unlikely for several more years. Any flow-on interactions to other members of the food web should take five or more years as they require substantial changes in abundance or size-distribution of the keystone commercial species before they are manifest.

In the Tasmanian MPA study, for example, rock lobster biomass increased an order of magnitude over a five year period, and it was only at that time when flow-on effects to populations of large grazing invertebrates developed. Densities of sea urchins, abalone and grazing gastropods all showed major declines in population numbers from the five year mark (Buxton *et al.* 2005), with population decline of these invertebrate species showing no sign of abating in recent monitoring surveys 15 years post protection. Outcomes after two years have been similar in both the Jurien Bay and NSW Jervis Bay marine parks, with few observable changes at that time. Significant increases in densities of large fishes and the targeted species *Cheilodactylus fuscus* were, however, evident in Jervis Bay sanctuary zones after four years (Edgar and Barrett, unpublished data). Ecosystem level changes related to increasing predator numbers, decreasing grazing invertebrate numbers, and increasing macroalgal cover required ca. 20 years to manifest in the NZ Leigh Marine Reserve (Babcock *et al.* 1999; Shears and Babcock 2002; Shears and Babcock 2003).

Because so little time has elapsed in relation to the life-spans of the large predatory species, it is not yet possible to adequately assess whether the JBMP management plan is on course to achieve its vision (Department of Conservation and Land Management 2005): “In the year 2025, the marine flora and fauna, habitats and water quality of the Jurien Bay Marine Park will be in the same or better condition than in the year 2005. The area will support viable and ecologically sustainable fishing, aquaculture, recreation and nature-based tourism and the marine park will be considered an important asset by the local community”.

In the context of this vision, it is important to recognise that a lack of difference through time in sanctuary and scientific reference zones compared to general use zones does not necessarily represent a failure of management aims with respect to biodiversity conservation. Some benefits of protected zones may propagate across the wider JBMP region through

dispersal of eggs, larvae, juveniles and commercial stock, increasing population numbers across all zones.

Nor does a lack of change through time in protected zones relative to baseline conditions represent a failure of management aims, providing that local ecosystems were not heavily fished prior to enactment of fishing restrictions. Fishing pressure on inshore WA coastal waters will inevitably increase with time, hence the safeguarding of areas with little current fishing pressure provides important insurance for local marine biodiversity through the long term.

In fact, protection of habitats that have been little unaffected by human activity to date is arguably a better strategy than the protection of habitats that are heavily-exploited and in need of restoration. Degraded habitats will not necessarily recover once exploitation ceases (Frank *et al.* 2005), and the social and political cost of protecting exploited habitats may be high because of the number of stakeholders with an interest in continued exploitation. Regardless, a fully representative system of MPAs will inevitably require protection of particular habitats that are heavily-exploited because some habitat types are currently exploited across their full distribution. Such habitats are most threatened and consequently in most need of protection if species extinction is to be avoided (Edgar *et al.* 2008b).

Recommendations

Our primary recommendation is that some locations with offshore reef need to protected as sanctuary zones. The lack of water depths >8 m in sanctuary zones clearly comprises a major deficiency in the JBMP zoning system. From a biodiversity perspective, it is, however, fortunate that numerous sanctuary zones are located in inshore waters, given that offshore reefs possess communities that are largely homogeneous across the full range of the JBMP. By contrast, inshore reef communities vary greatly between locations, and thus have a greater range of ecosystem biodiversity to be captured within the sanctuary zone system.

Regardless, neither biodiversity nor scientific aims of the JBMP can be fully achieved without the addition of offshore habitat types where rock lobsters as well as fishes are protected. Biodiversity aims will not be achieved because of the life-cycle of rock lobsters, which includes an offshore migration for most of the population at about minimum legal size (Phillips 1983), and also because of the potentially keystone role they play in controlling populations of grazers in local ecosystems (Edgar 1990). As a consequence of their offshore migration, no habitat will contain numbers of mature animals that approach unexploited levels, and thus no habitat exists where the full predatory impact of adult rock lobsters is expressed. This impact is likely to be considerable given the high densities at which western rock lobsters occur, and, by analogy, because of the considerable role played by rock lobsters elsewhere (Barkai and Branch 1988; Pederson *et al.* 2008; Shears and Babcock 2002; Tegner and Dayton 1999; Tegner and Dayton 2000; Tegner and Levin 1983).

Scientific aims of the JBMP are also presently compromised by the lack of offshore sanctuary zones because the role of rock lobsters in local ecosystems cannot be fully assessed, nor can data useful for rock lobster management be directly obtained from a population containing mature animals at natural densities, including data on natural growth and mortality of large individuals. The difficulty in undertaking an adequate scientific evaluation of effects of rock lobster harvesting has multiple consequences for the fishery, including relevance to the

granting of export permits for rock lobster exports under the Environmental Protection and Biodiversity Conservation Act.

We recommend that sanctuary zones be extended to the outer reef area when management zone boundaries are reviewed. Extra sites should then be added to the monitoring program to allow improved assessment of effects of fishing restrictions, particularly with respect to assessment of the ecosystem role of rock lobsters.

The population of Jurien Bay has steadily increased over the last decade and the imminent opening of the "Indian Ocean Drive" connecting Lancelin with Cervantes will increase access to the Midwest, with accompanying increase in recreational activities. As fishing pressure increases within the region, stakeholder groups may suggest the opening up of protected zones to fishing. We recommend that such suggestions be rejected for the reasons described in Section 4.3—management aims can be achieved despite little ecological change through time in protected zones, providing that initial fishing pressure was low. In fact, increasing pressure to open up protected zones to fishing implies a perceived desire amongst fishers to access grounds because of declining stocks in fished areas. In this case, opening up of protected zones to fishing will have major negative consequences for biodiversity. Note that a decline in stocks in fished areas should be apparent in the reef monitoring data in the form of a decline in population numbers of exploited species in general use zones.

Any change to the protection status of scientific reference and sanctuary zones would negatively affect the long-term JBMP reef monitoring program, with reduction in site replication for data extending from the 2004 baseline. Changes in the zoning scheme would also affect analysis of long-term change across the region for reef communities, including identification and assessment of ecological impacts of climate change.

We recommend that, other than for an extension of sanctuary zones to the offshore region, the system of protected zones in the JBMP be maintained with as few changes to regulations and zone boundaries as possible through the long term. This is particularly important for zones that include sites surveyed as part of the long-term JBMP reef monitoring program.

The JBMP reef monitoring program has been underway for only two years since the enforcement of restrictions on fishing, a period that is insufficient to adequately assess ecological changes associated with the zoning scheme. While changes in populations of exploited species are expected to occur most rapidly over the first five years of the zoning scheme, flow-on effects amongst other ecosystem components may take decades to become apparent, with relatively little change between years.

Given the high cost of JBMP field surveys (ca $25,000 per event), the frequency of surveys requires ongoing review to maximise return on expenditure. Thus, while annual surveys of fishes and invertebrates are appropriate over the initial period of most rapid ecological change, annual surveys of fishes and macroinvertebrates may not be necessary once annual change stabilises. The frequency of surveys of macroalgae and sessile invertebrates has already been reduced to once every two years because of the extra time, specialised skills and cost required to survey this component of the biota, and because changes in macroalgae associated with JMBP zones are not predicted until after substantial change occurs in populations of exploited habitat-engineering species.

Although the JBMP reef monitoring program was primarily designed to detect ecological changes associated with enactment of the marine park, it is also important to recognise that it

will have increasing value through the long-term in the identification and assessment of ecological impacts of climate change (and perhaps also invasive species). Many ecological changes that follow changing climate will be interactive and unpredictable, hence a critical need will always exist for empirical data. This role of the JBMP reef monitoring program may well be seen to more than justify field survey costs in future years.

We recommend that surveys of fishes and mobile invertebrates be repeated on an annual basis, and surveys of plant assemblages be conducted on a biennial basis, for at least five years from the time of enforcement of fishing restrictions in 2005. The frequency of field surveys should be reviewed in 2010 to assess whether a longer period between surveys is warranted on grounds of cost-effectiveness. Monitoring should nevertheless continue through the longer term at least until biotic changes associated with MPA protection stabilise, probably longer given the unique value of the data set in tracking ecological effects of climate change on temperate reef communities.

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